CONSTRUCTED WETLANDS FOR SLUDGE TREATMENT

A sustainable technology for sludge management

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Prologue

Sludge management has become a key issue in municipal wastewater treatment. Due to the elevated water content of the sludge, large daily flow rates need to be handled and treated. Moreover, solid waste management and disposal are among the most complex problems of wastewater treatment facilities. The main sludge treatment operations are aimed to increase the concentration of total solids in order to reduce the sludge volume to be disposed of (i.e. sludge thickening and dewatering); and to decrease the concentration of volatile solids and stabilise the biodegradable fraction of organic matter (i.e. sludge stabilisation).

Sludge treatment wetlands, also known as sludge drying reed beds, are rather new sludge treatment systems based on constructed wetlands. Sludge treatment wetlands have been used in Europe for sludge dewatering and stabilisation since the late 1980s. The largest experience comes from Denmark, with over 140 full-scale facilities currently in operation. In Northern Europe, other systems are located in Poland, Belgium and the United Kingdom; whereas in the Mediterranean region full-scale systems are operating in Italy, France and Spain. The main advantages of sludge treatment wetlands include low energy requirements, reduced operating and maintenance costs, and integration in the environment. However, the number of plants in operation is still scarce compared to other (conventional) technologies.

This book gathers experiences on sludge treatments wetlands from different European countries. Design and operation criteria adopted in Europe are here summarised. Focus is put on describing dewatering and mineralisation processes, leading to final biosolids characteristics that makes them appropriate for land application as organic fertilisers and soil conditioners. Finally, economic and environmental aspects are addressed.

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Introduction

Alternativas de gestión de lodos en Cataluña

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En Catalunya operan 375 EDAR urbanas, que en el año 2009 trataron 675,6 hm^3 de agua residual. La depuración de aguas residuales urbanas genera un total de 548.224 t de biosólidos al año, con una sequedad promedio del 23,9%, en el año 2009. El 96% de los biosólidos generados son sometidos a tratamiento previamente a su destino final, el cual es, mayoritariamente, la valorización agronómica (un 79% del lodo generado en el 2009). El resto del lodo es destinado a valorización energética (19% del total) y a vertedero (2%).

En la actualidad se sanea alrededor del 95% de la población. Más de la mitad de las EDAR en servicio tienen un caudal de diseño inferior a 1.500 m^3/día y una tercera parte del total inferior a 400 m^3/día (equivalente a 2.000 habitantes equivalentes, aproximadamente). Más de la mitad de las EDAR de Catalunya (56%) generan menos de 50 t de biosólidos al año (en materia seca), lo que representa una producción inferior a 150 kg MS/día. Por otra parte más de 1/3 de las EDAR en servicio no disponen de sistemas de deshidratación y los biosólidos generados por las mismas son destinados a otras EDAR para su deshidratación.

EVOLUCIÓN DE LA PRODUCCIÓN

El Programa de Saneamiento de aguas urbanas de Catalunya prevé que al final del periodo de vigencia del mismo (año 2015) habrá en servicio alrededor de 1.500 sistemas de saneamiento. Así pues, en los próximos años está prevista la entrada en funcionamiento de más de 1.000 nuevas EDAR. Estas instalaciones se ubicarán mayoritariamente en núcleos alejados y de pequeña dimensión, los cuales no pueden ser cubiertos por la actual red de saneamiento en alta. Considerando una población pendiente de sanear aproximada de 400.000 habitantes[1], la dimensión promedio de las nuevas EDAR se situará en el entorno de los 400 habitantes. La naturaleza primordialmente doméstica de las poblaciones a sanear implicará una equivalencia entre la población equivalente y la población realmente servida.

En los últimos años se ha producido una estabilización de la producción de biosólidos en Catalunya, a pesar del incremento de la población que se ha experimentado. Concretamente, el año 2009 la generación ha disminuido en un 6%. La actual cobertura del saneamiento a la práctica totalidad de la población catalana permite estimar que el incremento de producción derivado de las nuevas EDAR se verá compensado por la disminución de producción derivada del continuo descenso del porcentaje de la carga industrial que reciben las EDAR urbanas y de la entrada en servicio de sistemas de estabilización del lodo (como sistemas de digestión). Así pues, durante los próximos años se prevé el mantenimiento de la generación de biosólidos con un ligero crecimiento del 0,7% anual.

[1] En base al censo actual que se ajustará según el crecimiento poblacional que se desarrolle.

PLANIFICACIÓN DE LA GESTIÓN DE BIOSÓLIDOS

Factores que condicionan la gestión de los biosólidos

La gestión de los biosólidos se encuentra sometida a factores que la condicionan. Estos factores son diversos y se encuentran en constante evolución. Los condicionantes a la gestión del lodo se pueden dividir según la siguiente clasificación:

1) Legales
2) Sociales
3) Ambientales
4) Energéticos
5) Técnicos
6) Logísticos
7) Económicos/Financieros
8) Otros

Se pone de relieve que la mayor parte de factores que afectan a la gestión de los biosólidos son de naturaleza dinámica y resulta difícil predecir su futura evolución.

Programa de lodos de Catalunya

La planificación de la gestión de los biosólidos en Catalunya se establece en el *Programa d'actuacions per a la gestió dels fangs residuals generats en els processos de depuració d'aigües residuals urbanes de Catalunya –Programa de lodos–*. Este programa, elaborado por en la *Agència Catalana de l'Aigua*, se encuentra en proceso de aprobación.

La actual gestión de biosólidos en Catalunya se considera sostenible en todos los sentidos. No obstante, esta gestión se encuentra sometida a diversos condicionantes de evolución incierta. En base a la anterior circunstancia, el *Programa de lodos* tiene como principales objetivos:

1) Reducir las incertidumbres que afectan a la gestión
2) Mejorar la actual gestión de los biosólìodos

Para conseguir estos objetivos el *Programa de lodos* establece los siguientes criterios de actuación:

- Diversificar la gestión para disponer de alternativas en caso de evolución de los factores que la condicionan
- Aplicar los criterios de proximidad y autosuficiencia en la gestión
- Aplicar la jerarquía de gestión de residuos: 1) Reducción; 2) Valorización; y 3) Disposición
- Reducción del impacto de la gestión

Estos criterios se concretan en las siguientes acciones:

1) Implantación de nuevas digestiones anaerobias en EDAR
2) Minimización del destino a vertedero
3) Implantación de nuevos compostajes
4) Ampliación de secados térmicos
5) Valorización energética del lodo seco
6) Continuidad en la aplicación a la agricultura de lodo deshidratado
7) Continuidad en la utilización de la capacidad de compostaje privada
8) Investigación en mejora y nuevas vías de gestión

LECHOS DE MACRÓFITOS PARA EL TRATAMIENTO DE BIOSÓLIDOS

La utilización de humedales para el tratamiento de biosólidos es una práctica validada científica y experimentalmente. El sistema de tratamiento se basa en la utilización de lechos de macrófitos para estabilizar y deshidratar el lodo. En Catalunya, existen diversas EDARs donde se ha implantado este tipo de tecnología para el tratamiento de la totalidad de la producción de lodo de las mismas.

El sistema presenta diversas ventajas frente a otro tipo de vías de gestión y permite satisfacer los objetivos de gestión del *Programa de lodos de Catalunya*, especialmente para EDARs de pequeña dimensión y alejadas de los centros de tratamiento de biosólidos. Como se ha comentado anteriormente, este tipo de EDAR representa una parte significativa de las instalaciones actualmente en servicio y constituirá la inmensa mayoría del parque de EDAR de próxima implantación.

La utilización de humedales permite un tratamiento de biosóliods *in situ*, consiguiendo la práctica ausencia de salida de residuos de la instalación durante periodos de tiempo prolongados (superiores a un año). Los análisis del producto resultante permiten considerarlo un lodo tratado, con un notable descenso del contenido del agua respecto a la entrada y presentando valores de estabilidad equivalentes, como mínimo, a un lodo digerido aneròbicamente.

Las ventajas de la utilización de lechos de macrófitos en EDAR de pequeña dimensión se puede resumir en el siguiente listado:

1) Aplicación estricta del criterio de proximidad y autosuficiencia en la gestión de residuos, los cuales se tratan en el mismo punto de generación.
2) Aplicación de la jerarquía de gestión de residuos: prevención de la producción de biosólidos (en materia seca y en materia total) y valorización del producto resultante.
3) Ausencia de impactos en la gestión, en reducirse el número y la frecuencia de las operaciones de carga y descarga y de transporte del residuo.
4) Posible habilitación de la aplicación directa del biosólido en provecho de la agricultura, en poderse considerar el lodo como tratado.
5) Ahorro económico en sustituirse las operaciones de transporte y tratamiento del lodo líquido. Este ahorro es importante en función de la dimensión de la planta y de la distancia a los centros de tratamiento (por ejemplo otras EDAR).

Complementariamente, la mejora de la gestión permite dar una respuesta positiva a los condicionantes que afectan a la gestión de los biosólidos, especialmente los de carácter económico, logístico, ambiental y social. Al tratarse de una solución de tipo local se favorece la corresponsabilización de la población servida y se minimiza un eventual rechazo social.

En estas circunstancias, se consiguen cumplir los objetivos del *Programa de lodos de Catalunya*, mejorando la gestión del residuo y reduciendo las incertidumbres de gestión del biosólido al minimizar la producción, estabilizar el material, reducir el impacto ambiental y social de su gestión.

En cuanto a los criterios de gestión, la implantación de humedales permite diversificar la gestión del lodo, ampliándose las vías para su destilación final; se aplican, como se ha mencionado, los criterios de autosuficiencia y proximidad en la gestión; se aplica la jerarquía de gestión de residuos; y se reduce el impacto de la gestión.

General aspects of sludge composting

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THE COMPOSTING PROCESS

Although there is no universally accepted definition of composting, Haug (1993) uses a practical definition of the process, which provides all the main points to obtain a successful process: “Composting is the biological decomposition and stabilization of organic substrates, under conditions that allow development of thermophilic temperatures as a result of biologically produced heat, to produce a final product that is stable, free of pathogens and plant seeds, and can be beneficially applied to land.”

This practical definition is extremely useful for scientists, technicians and specially composting plant managers, since it states the three main characteristics of the process:

1) It is a biological aerobic process: this means, on the one hand, that the process conditions must be adequate for the development of microbial communities, although compost microbiology has not been studied in detail because of the inherent difficulty of cultivating microorganisms coming from heterogeneous solid samples. However, in recent publications, new microbial techniques have been successfully applied to the identification of viable compost strains (Amir *et al.* 2008). On the other hand, composting is aerobic. This implies that oxygen must be effectively transferred from air to the cells and carbon dioxide must be transported from cells to exhaust air. Several methods have been proposed in different technologies to provide oxygen to composting materials. In the passive aeration method, oxygen supply is achieved by means of the natural convective movement of the air through the pile (Mason *et al*. 2004). To achieve this, the size and porosity of the pile should be adequate to enable aeration (Szanto *et al*. 2007). Turned composting systems are passively aerated but additional turning is used to maintain the proper porosity, to provide oxygen, to mix the material and to release excessive heat, water vapour and carbon dioxide. In static forced-aerated pile composting, forced aeration is applied by means of air ducts, and aeration is provided by blowing or sucking air through the composting material, which must present an adequate level of porosity (Haug 1993).

2) Composting deals with organic solid wastes: this is especially important because of the inherent heterogeneity of organic substrates, which causes that sampling in composting processes is not a simple task (Barrena *et al.* 2006a). Also, the existence of several phases (gas, liquid and solid) implies that transfer mechanism for both mass and energy can limit the overall process rate. In the case of mass transfer, oxygen diffusion is the key issue (Scaglia *et al*. 2000), whereas energy is, in general, poorly transferred through organic matter (Barrena *et al*. 2006b), causing the self-heating of compost that eventually results in material sanitation and stabilization.

3) Compost must be sanitized and stable: compost application must be carried out in a safe manner. This implies that proper conditions of sanitation must be ensured during the process. In fact, it is considered that the high temperature reached due to the metabolic heat generated during the thermophilic phase of the composting process is effective in destroying the pathogens (Wong and Fang 2000). To regulate this point, several recipes have been proposed to ensure compost sanitation. For instance, in sludge composting, different combinations of temperature and time are indicated in order to reach the proper disinfection of the final product (temperatures over 55°C, 20 days for conventional aerobic treatments or 20 hrs for 55°C for advanced aerobic stabilization treatments) (European Commission 2000). Other international rules on sludge disinfection by composting propose similar time–temperature conditions (US Environmental Protection Agency 1995). In the animal by-products category, European legislation describes the exact sanitation conditions to be ensured for a proper composting process (Regulation (EC) No 1774/2002). In relation to biological stability, this is an important issue for composting process performance and obviously for compost quality.

COMPOSTING THECNOLOGY

Haug (1993) gives an extended description of the composting systems developed at that moment. Although new systems have been implemented since then, from a practical point of view, composting plants for the processing of organic solid wastes can be still divided into two main categories:

1) Piles or windrows: for rural or semi-rural areas, usually plants with a capacity range of 1,000-40,000 metric tons/year. This traditional composting method was implemented in the first composting plants constructed in the world, and it is based on the use of mechanically turned or forced-aerated static piles. Typically, temperature, moisture and oxygen content inside the material are monitored during the first weeks of composting (thermophylic initial phase), and weekly during the curing phase (mesophylic final phase). Total composting time is usually about 12-13 weeks.

2) In-vessel or reactor systems: for urban and high-density population areas, these plants have a capacity range of 10,000-100,000 metric tons/year. In this case, material remains for few weeks in a digester (tunnels are the most popular) with forced aeration systems and on-line monitoring of temperature, oxygen, carbon dioxide and ammonia exhaust gases. Data from different probes are computer collected and some control recipes can be applied to the system, usually in the form of temperature and oxygen setpoints, allowing a rapid decomposition of organic matter. Afterwards, material is piled for the curing phase for 5-6 weeks. Typically, in-vessel plants are composed of the composting reactors and the gas collection, transportation and cleaning units (being biofilters the most popular).

Of course, there exist a lot of modifications and variations of these two main processes, whose suitability mainly depends on the properties of the feedstock to be composted. The study of the performance of these plants according to the evolution of global biological activity indicators has been the main objective of some recent research works (Barrena *et al*. 2008; Ponsá *et al*. 2008; Ruggieri *et al*. 2008).

COMPOSTING AS BIOLOGICAL PROCESS. PARAMETERS AFFECTING BIOLOGICAL ACTIVITY

In order to control and optimize the bio-kinetics of the composting process to produce a compost of desired quality, it is important to understand the factors that influence the process. A composting matrix is an ecosystem of interdependent interactions between biotic and abiotic factors that cause degradation of organic matter. The abiotic and biotic factors playing key role in the composting process (Pietronave *et al.* 2004; Gajalakshmi and Abbasi 2008) are described next.

Abiotic factors

- Nature of the substrate: Several kinds of organic residues susceptible to the enzymatic activities of the microorganisms can be converted into compost if necessary conditions for biodegradation are provided. As the substrate becomes the only source of food to the microorganisms in a composting matrix, the nature of the substrates is the most controlling factor in any composting process (Gajalakshmi and Abbasi 2008). The organic compounds in biowastes could be hence classified into three main categories (Komilis *et al*. 2004; Gajalakshmi and Abbasi 2008): (1) carbohydrates (polymers and simple sugars), (2) lignin, and (3) nitrogen compounds. In the beginning of the composting process, simple carbohydrates are converted to carbon dioxide and water (Bernal *et al.* 1998), and degradation of nitrogenous compounds results mainly in ammonia volatilization. In the later stages of composting, cellulose and hemicellulose are utilized by the compost microflora and eventually lignin is also subjected to slow degradation. Besides mineralization, organic matter is converted to humic substances (Quagliotto *et al.* 2006).
- Carbon/Nitrogen ratio: The relative proportion of carbon and nitrogen is also a major controlling factor in the composting process (Agnew and Leonard 2003). Carbon serves primarily as an energy source for the microorganisms, while a small fraction of the carbon is incorporated to the microbial cells. Nitrogen is critical for microbial population growth (Gajalakshmi and Abbasi 2008). If nitrogen is limiting, microbial populations will remain small and decomposition rates for available carbon will be lower. Excess nitrogen is lost from the system as ammonia gas (de Guardia *et al.* 2008). According to Golueke (1992), rapid and entire humification of substrates by the microorganisms primarily depends on it initially having a C/N ratio between 25 and 35. Anyway, it must be noted that the biodegradable C/N ratio can be significantly different from typical C/N chemically determined (Sánchez 2007).
- Moisture: Moisture is one of the composting variables that affects microbial activities to a considerable extent since it provides a medium for the transport of dissolved nutrients (Hamelers 2004) required for the metabolic and physiological activities of microorganisms (Agnew and Leonard 2003).
- Oxygen, temperature and aeration interaction: The microbial decomposition process enhances the interdependence and mutual control between two of the main composting parameters, oxygen levels and temperature. The temperature within a composting matrix determines the rate at which many of the biological processes take place (Agnew and Leonard 2003) and controls the development and the succession of the microbiological flora (Taiwo and Oso 2004). A temperature in the range of 55 to 65°C allows for considerable destruction of pathogenic organisms (Smith *et al.* 2005).
- pH: pH also significantly affects the composting process. The range of pH values suitable for bacterial development is 6.0-7.5, while fungi prefer an environment in the range of pH 5.5-8.0 (Zorpas *et al.* 2003).

Biotic factors

Composting involves a myriad of microorganisms. The composition and magnitude of these microorganisms are important components of the composting process. The microbes decompose the organic matter, and transform the nitrogen component through oxidation, nitrification, and denitrification. Bacteria play the dominant role during the most active stages of composting process because of their ability to grow rapidly on soluble proteins and other readily available substrates. Strom (1985b) reports that as much as 87% of the randomly selected colonies during the thermophilic phase of composting belong to the genus *Bacillus*. The role of fungi starts when simple, easily degradable substances such as sugar, starch, and protein are acted upon by bacteria and the substrate is predominated by cellulose and lignin, which normally occurs toward the curing stage of the composting process (Tiquia *et al.* 2002). Most fungi are eliminated by high temperatures, but they commonly recover when temperatures are moderate (Tiquia *et al.* 2001), and the remaining substrates are predominantly cellulose or lignin (Bertoldi and Vallini 1983). Like fungi, actinomycetes also utilize complex organic material. They tend to grow in numbers in the later stages of composting, and have been shown to attack polymers such as hemicellulose, lignin, and cellulose (Bertoldi and Vallini 1983). Actinomycetes are able to degrade some cellulose and hydrolyse lignin, and are tolerant of higher temperatures and pH than fungi. Thus, actinomycetes are important for lignocellulosic degradation during peak heating. Actinomycetes are thus well adapted to exploit the compost environment as the piles cool in the immediate post peak heat phase.

Different microbial communities predominate during the various composting phases, each of which being adapted to a particular environment. Primary decomposers create a physico-chemical environment suited for secondary organisms, which cannot attack the initial substrates, while metabolites produced by the one group can be utilized by the other. The initial rapid increase of temperature involves a rapid transition from mesophilic to thermophilic microflora (Ryckeboer *et al.* 2003a). Often a disruption of the process is observed at temperatures between 42 and 45°C. The initial mesophilic microflora is inhibited by the high temperature, while the thermophilic populations have not yet developed and are below their temperature optimum.

Only when a sufficient number of thermophiles is generated, temperatures rise again. At temperatures exceeding 60°C, the optimum for most thermophiles is reached, and the system starts to limit itself due to the inhibitory high temperatures. Heat may in principle inhibit organisms through enzyme inactivation (Gajalakshmi and Abbasi 2008) or may limit oxygen supply. An efficient process kinetics control thereupon provided through regular aeration, the thermophilic stage continues until the heat production becomes lower than the heat dissipation, due to the exhaustion of easily degradable substrates. High temperatures support degradation of recalcitrant organics and elimination of pathogenic and allergenic microorganisms. During the second mesophilic (cooling) phase nutrients become a limiting factor, causing a decline in microbial activity and heat output. During the maturation phase, the substrate quality further declines and compounds such as lignin-humus complexes are formed that are not further degradable. As corollary, the inherent complexity of substrates and intermediate biochemical reactions and their products, make the microbial diversity and the succession of populations vital in ensuring an efficient bio-kinetic process control and biodegradation during the composting process.

REFERENCES

Agnew JM, Leonard JJ (2003) Literature review – the physical properties of compost. *Compost Science and Utilization* 11, 238-264

Amir S, Merlina G, Pinelli E, Winterton P, Revel JC, Hafidi M (2008) Microbial community dynamics during composting of sewage sludge and straw studied through phospholipid and neutral lipid analysis. *Journal of Hazardous Materials* 159, 593-601

Barrena R, Pagans E, Faltys G, Sánchez A (2006a) Effect of inoculation dosing on the composting of source-selected organic fraction of municipal solid wastes. *Journal of Chemical Technology and Biotechnology* 81, 420-425

Barrena R, Cánovas C, Sánchez A (2006b) Prediction of temperature and thermal inertia effect in the maturation stage and stockpiling of a large composting mass. *Waste Management* 26, 953-959

Barrena R, Vázquez F, Sánchez A (2008) Dehydrogenase activity as a method for monitoring the composting process. *Bioresource Technology* 99, 905-908

Bernal MP, Sánchez-Monedero MA, Paredes C, Roig A (1998) Carbon mineralization from organic wastes at different composting stages during their incubation with soil. *Agriculture, Ecosystems and Environment* 69, 175-180

Bertoldi M de, Vallini PA (1983) The biology of composting: A review. *Waste Management Research* 1, 157-176

De Guardia A, Petiot C, Rogeau D, Druilhe C (2008) Influence of aeration rate on nitrogen dynamics during composting. *Waste Management* 28, 575-587

European Commission (2000) Working document on sludge 3rd draft. Available online: http://ec.europa.eu/environment/waste/sludge/pdf/sludge_en.pdf

Gajalakshmi S, Abbasi S (2008) Solid waste management by composting: State of the art. *Critical Reviews in Environmental Science and Technology* 38, 311-400

Golueke CG (1992) Bacteriology of composting. *Biocycle* 33, 55-57

Hamelers HVM (2004) Modeling composting kinetics: A review of approaches. *Reviews in Environmental Science and Biotechnology* 3, 331-342

Haug RT (1993) *The Practical Handbook of Compost Engineering*, Lewis Publishers, Boca Raton, Florida, USA, pp 35-65

Komilis DP, Ham RK, Park JK (2004) Emission of volatile organic compounds during composting of municipal solid wastes. *Water Research* 38, 1707-1714

Mason IG, Mollah MS, Zhong MF, Manderson GJ (2004) Composting high moisture content bovine manure using passive aeration. *Compost Science and Utilization* 12, 249-267

Szanto GL, Hamelers HVM, Rulkens W, Veeken A (2007) NH_3, N_2O and CH_4 emissions during passively aerated composting of straw-rich pig manure. *Bioresource Technology* 98, 2659-2670

Pietronave S, Fracchia L, Rinaldi M, Martinotti MG (2004) Influence of biotic and abiotic factors on human pathogens in a finished compost. *Water Research* 38, 1963-1970

Ponsá S, Gea T, Alerm L, Cerezo J, Sánchez A (2008) Comparison of aerobic and anaerobic stability indices through a MSW biological treatment process. *Waste Management* 28, 2735-2742

Quagliotto P, Montoneri E, Tambone F, Adani F, Gobetto R, Viscardi G (2006) Chemicals from wastes: Compost-derived humic acid-like matter as surfactant. *Environmental Science and Technology* 40, 1686-1692

Ryckeboer J, Mergaert J, Coosemans J, Deprins K, Swings J (2003a) Microbiological aspects of biowaste during composting in a monitored compost bin. *Journal of Applied Microbiology* 94, 127-137

Ruggieri L, Gea T, Mompeó M, Sayara T, Sánchez A (2008) Performance of different systems for the composting of the source-selected organic fraction of municipal solid waste. *Biosystems Engineering* 101, 78-86

Sánchez A (2007) Discussion: A kinetic analysis of solid waste composting at optimal conditions. *Waste Management* 27, 854-855

Scaglia B, Tambone F, Genevini PL, Adani F (2000) Respiration index determination: Dynamic and static approaches. *Compost Science and Utilization* 8, 90-99

Smith SR, Lang NL, Cheung KHM, Spanoudaki K (2005) Factors controlling pathogen destruction during anaerobic digestion of biowastes. *Waste Management* 25, 417-425

Strom PF (1985b) Identification of thermophilic bacteria in solid waste composting. *Applied Environmental Microbiology* 50, 907-913

Taiwo LB, Oso BA (2004) Influence of composting techniques on microbial succession, temperature and pH in a composting municipal solid waste. *African Journal of Biotechnology* 3, 239-243

Tiquia SM, Wan JHC, Tam NFY (2001) Extracellular enzyme profiles during co-composting of poultry manure and yard trimmings. *Process Biochemistry* 36, 813-820

Tiquia SM, Wan JHC, Tam NFY (2002) Microbial population dynamics and enzyme activities during composting. *Compost Science and Utilization* 10, 150-161

US Environmental Protection Agency (1995) A Guide to the Biosolids Risk Assessments for the EPA Part 503 Rule. Available online: http://www.epa.gov/OW-OWM.html/mtb/biosolids/503rule/index.htm

Wong JWC, Fang M (2000) Effects of lime addition on sewage sludge composting process. *Water Research* 34, 3691-3698

Zorpas AA, Arapoglou D, Panagiotis K (2003) Waste paper and clinoptilolite as a bulking material with dewatered anaerobically stabilized primary sewage sludge (DASPSS) for compost production. *Waste Management* 23, 27-35

Sludge dewatering and mineralization in reed beds: Design and operation considerations

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Abstract
Planted reed beds are widely used in Northern Europe to dewater and mineralise surplus sludge from activated sludge systems treating urban domestic sewage. Although in theory a simple technique relying largely on natural processes, experience has shown, that it is very important to understand and respect the basic design and operation requirements. This paper describes the basic design and operation requirements of planted reed beds, with special focus on pivotal requirements to respect to secure proper functioning. When these requirements are not respected, the systems are likely to fail. Commonly observed problems are related to wrong construction, and usually in concert with inappropriate operation. The number of beds must be sufficient to secure long resting periods, and the bed filter must have a high water permeability and an efficient passive aeration of the drainage system. Operation of the system must entail a running-in period for plant establishing, and the loading strategy should include long resting periods of the individual beds for sludge conditioning. A vigorous growth of reeds in the beds is a key to success, as the plants are responsible for keeping a sufficient hydraulic conductivity and also for removing capillary water from the sludge through transpiration. Not all types of sludge can be processed in planted reed beds. Sludge that are not stabilised and sludge that traditionally are difficult to dewater are also problematic in planted reed beds. Extreme care should be taken when attempting to extrapolate the use of planted reed beds to applications and regions outside of its 'normal' and documented area of application. However, there is no doubt that planted reed beds have a great application potential in warm and dry climatic regions.

Keywords
Activated sludge; evapotranspiration; *Phragmites australis*; sludge dewatering

INTRODUCTION

Surplus sludge generated from biological and chemical wastewater treatment needs further management before it can be finally disposed or used as an agricultural resource. Traditional sludge dewatering practices reduce its volume, increase its dry matter content, and consequently minimise transportation and management costs. Several successful and well-documented methods are available, but their capacity and operation vary, as well as the level of technological sophistication, infrastructure requirements, and needs for operational labour skills. Centrifuges and belt filter presses generally require the addition of conditioning chemicals (e.g. coagulants and/or polyelectrolyte) and a significant input of energy. Planted reed beds, on the other hand, are low technology, energy efficient and do not require addition of chemicals. They dewater and stabilize the sludge and produce a final product that can be safely disposed or used for agricultural purposes. An additional advantage of planted reed beds as compared to most other technologies is that the water released from the sludge during dewatering is treated as it percolates through the bed. The typically high concentrations of COD and BOD in the reject water are reduced by more than 60%, and ammonium-N is usually nitrified. In addition, long-term sludge reduction takes place in the reed beds due to dewatering and mineralisation of the organic matter in the sludge.

Planted reed beds have been used in Europe for sludge dewatering and stabilization since the late 1980s. The largest experience comes from Denmark, where there are +100 full-scale systems in operation. Experiences from these show that planted reed beds are capable of treating many types of sludge with a dry solid content between 0.5 and 5%. Dimensioning and design of the reed bed systems depend on the sludge production (tonnes of dry solids/year), sludge type, quality and regional climate. The operation of a system is divided into a number of phases related to different periods in the lifetime of a system. The phases comprise the periods of commissioning, full operation, emptying and re-establishment of the system. Loading cycles are related to the sludge type and the age of the sludge reed systems. The sludge residue will, after approximately 10 years of operation, reach an approximate dry solids content of 30%. Experience has shown that the quality of the final product with respect to heavy metals, hazardous organic compounds and pathogens after 10 years of treatment make it possible to recycle the sludge residue to agriculture.

PLANTED REED BEDS

System design

A planted reed bed system for sludge dewatering generally consists of a number of reed beds (usually +8) that are loaded in sequence by liquid sludge from the wastewater treatment plant. The reed beds are built as planted vertical filters with an efficient drainage system in order to dewater the sludge effectively. The individual reed bed is usually established in a plastic lined excavation in the ground (Fig. 1) or with vertical concrete sidewalls on top of the soil, depending on site conditions. Walls should have enough freeboard (1.2 to 1.5 m) to stock up the sludge for a period of 10 years. In the bottom of the bed a dense system of drainage pipes are placed in a layer of coarse gravel. The drainage pipes are vented to the atmosphere to secure air exchange in the pipes and hence and effective transfer of oxygen from the atmosphere to the drainage layer. On top of the drainage layer, one to three finer textured layers of gravel, sand or soil are placed to filter the water from the sludge that is pumped onto the surface of the beds. The upper so-called ‘growth layer’ is planted with common reed (*Phragmites australis*). The sludge is usually loaded onto the surface of the beds via vertical standpipes (Fig. 2) with feeding lines buried in the drainage layer. Each bed requires emptying after a period of operation of about 10 years, after which the bed can be reloaded.

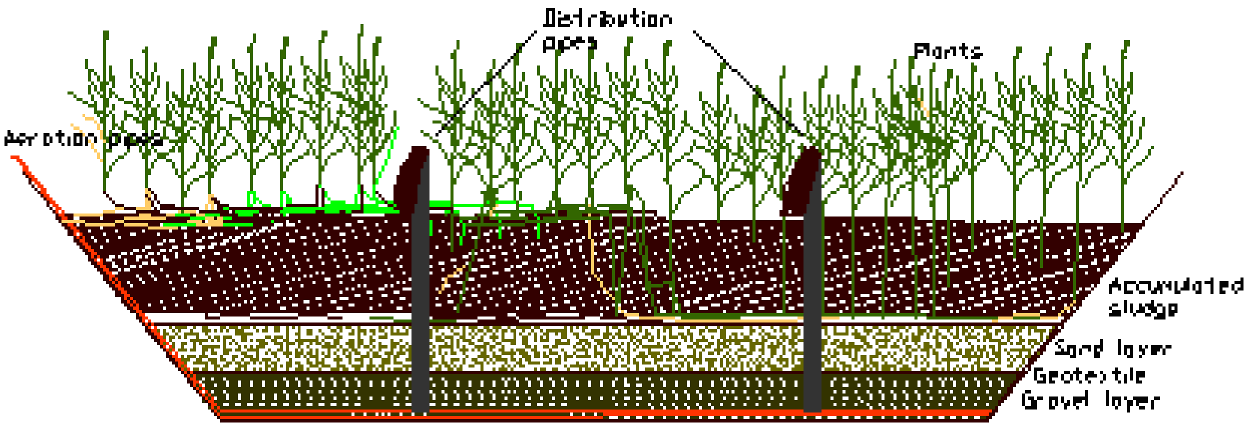

Figure 1. Diagram of a typical reed bed. The bed includes an impermeable basin to host the bed, plants, several layers of specific types of gravel, sand and soil, a distribution system with vertical standpipes, and a drainage system with efficient air-change via ventilation pipes extending into the atmosphere.

System operation

Any system should have several (minimum eight) beds to alternate the loading and provide enough time between loadings for the biological and physical processes to take place during the resting periods. During operation, the sludge from the wastewater treatment plant (usually surplus activated sludge) is pumped through the vertical standpipes and distributed passively across the bed surface (Fig. 2). The actual loading has to be relatively fast in order to secure a good distribution of the liquid sludge across the entire bed surface. The majority of the water percolates through the vertical filter and returns to the wastewater treatment system during the following hours/day. The solid fraction of the sludge stays on the surface of the bed. After each load, a dewatering period is allowed before a new layer of sludge is discharged on top of the dewatered sludge. The bed will then enter into a relatively long resting period where the sludge layer is allowed to dry. Oxygen diffusion via the drainage layer from below and through the cracked sludge surface into the sludge residue enables aerobic micro organisms to exist in the sludge residue. This process continues until the bed is filled with dewatered sludge and has to be emptied (after about 10 years). The water drained from the sludge percolates through the sand and gravel. The prevailing oxic conditions in the non-saturated filter and the filtering effect of the media reduce the concentration of pollutants in the released water, which is sent back to the wastewater treatment plant for treatment.

Figure 2. Vertical standpipes through which the liquid sludge is distributed over the entire surface of the bed. The picture shows a situation where a planted reed bed, after emptying, are being loaded, and plants are regenerating from the rhizomes remaining in the growth layer

There are three periods in the operation of a planted reed bed system. During start-up (about two years), sludge loading should be less than the designed loading rate. After start-up, the plants are fully developed and the bed can be loaded with the designed loading rate (usually about 60 kg dry matter per square metre per year). In the third period (after about eight years of operation), the beds are emptied to remove accumulated dewatered sludge. The beds that need emptying (maybe two out of eight) will not be loaded during the (dry) summer period to maximise dry matter content of the sludge. The beds are then emptied successively; depending on the needs and number of beds, it will take about four years. Once the bed is empty, a new start-up period begins.

Plant functions

The plants in the reed beds have several important functions. The root system and the stems help keep the filter open so that water can drain from the sludge to the drainage system. The growth of the rhizomes are important in this respect, as well as the wind induced movements of the stems which results in cracks and openings in their periphery that will allow the water to penetrate into the filter (Fig. 3). Another important function of the plants in to remove capillary bound water in the sludge by plant uptake and transpiration. The evapotranspiration rate of reeds is very high and can be even higher than the potential evaporation from an open water surface. This is very important during the resting phase, where capillary bound water in the sludge will be transpired, and as a result the sludge will tighten and craze. The cracks created are very important as these will allow easy transfer of air (and hence oxygen) into the sludge, that will stimulate mineralisation and further dewatering. The evapotranspiration is particularly important during the final resting period (usually several months during summer) before the beds have to be emptied. Because of the high evapotranspiration rates of the reeds which exploit the capillary bound water in the sludge, the dry matter content of the sludge may be as high as 40% when the beds are emptied (more commonly the final dry matter content is around 30%). The plants may have other functions, such as stimulation of microbial activities in the sludge, release of root exudates that interact with the sludge, and transfer of oxygen to the sludge through root release, but these functions are less documented and are probably of less important. Oxygen diffusion from the roots into the sludge residue might enable aerobic micro organisms to exist close to the roots and in the sludge residue.

Figure 3. Openings in the sludge around the stems of *Phragmites australis* caused by the movements of the plants in the wind. These openings are important for maintaining a high water permeability in the sludge layer accumulated on the surface of the bed.

DESIGN AND OPERATIONS CONSIDERATIONS

As always when a new technology is introduced into the market, some operational problems are likely to be observed in the first full-scale systems. Furthermore, some engineering design offices are likely to adopt and sell the system without proper knowledge of the critical design and operation issues. This has been the case in Denmark, and has led to the situation that several systems have had serious problems. The common problems observed include poor growth of reeds after one or two years, in worst cases no reeds at all, and liquid anaerobic sludge with a high water content in the beds (fig. 4). This situation is associated with inefficient drainage of the water from the bed. The dry matter content of the sludge will be only 10-12%, even after a resting period of several months. In the following, the main reasons for such problems will be described.

Figure 4. Typical planted reed beds with operational problems as indicated by the reduced and clumped growth of the reeds.

Design and construction issues

Low amount of beds: A number of systems have been designed with only 2 to 6 beds. This has shown to be problematic as the resting periods of the individual bed then are relatively short. The problem is compounded during the period of sludge emptying, where the beds that are going to be emptied have to be taken out of operation for an extended period.

Damaged drainage system: The drainage pipes are likely to be damaged if – during the construction phase – heavy machinery is used within the beds. This may result in the breakage and/or compression of the drainage pipes with resulting inefficient drainage. Also the levelling or inclination of the drainage system is important. The drainage system must be laid down with a slight slope to secure efficient runoff of drainage water. If levelling is not correct and some parts of the drainage pipes are continuously filled with water (e.g. at low spots), there will be no possibility for air ventilation of the drainage pipes, and the oxygen transfer to the drainage layer will be hindered. Systems have also been designed and constructed with standing water in the drainage layer, which evidently has the same effect.

Wrong composition of the growth layer: The upper layer of the filter material, the growth layer, is meant to support the growth of the plants, but at the same time the layer must have a composition so that water can easily drain through the layer. The contents of clay and fine silt-particles must therefore be low, and also the content of organic matter must be restricted. The texture of the material should be characterised by a high degree of uniformity (a steep grain-size distribution curve) to secure high water permeability. If the composition of the growth layer is wrong (and maybe the layer compacted from the use of heavy machinery), the water from the sludge will not be able to drain from the bed at a sufficiently high speed. The bed will remain wet and the sludge will not be dewatered sufficiently.

Wrong planting technique: Planting of the systems are best done by potted seedlings at a density of approx. 4 pots per m^2 into the growth layer. After planting, sludge loading rate and frequency have to be fitted to water and fertilize the plants. This means a reduced capacity of the system the first two growth seasons. Some suppliers have instead of potted seedlings used 20 cm by 20 cm 'blocks' of root systems from natural reed stands, and have placed these blocks on top of – and not buried into – the growth layer. By using this planting technique, the suppliers argue that the beds can be loaded at full capacity from the beginning, without damaging the plants. The experience shows however, that although the reeds from the blocks will survive and look healthy in the beginning, the roots and rhizomes will not spread into the growth layer and the sludge surrounding the root-blocks to any significant extent. Hence, after a couple of years, these systems generally have serious operational problems.

Operation issues

Wrong running-in period: During running in, the main emphasis of the operation is to secure an efficient establishment of the reed. During the first year of operation, sludge loading rate and frequency should be carried out with the main purpose of watering and fertilizing the reed. Even the second year, the loading has to be reduced, as compared with full capacity, to stimulate plant growth and establishment. If sludge loading rate is too high during the running in period, and if the loading frequency does not secure sufficient water for the reeds, the plants will not establish well and will eventually disappear from the system once the sludge layer increase in thickness.

Overloading: The treatment capacity of a planted reed bed, usually given as kg of dry matter per square metre per year, is an important design parameter, and depend on several factors including climate and sludge type and quality. For economic reasons systems are usually designed to receive as much sludge as possible, without jeopardizing performance. If the systems are overloaded compared to their treatment capacity, this is likely to result in operational problems, with negative effects on the plants and hence dewatering and mineralisation.

Lack of resting phases: Some systems have been operated continuously with daily loadings on all beds. Even though water will drain from the beds, the sludge will not dry out, and after a period serious problems with insufficient water drainage will occur. A resting phase that is long enough, so that the sludge will dry out and crack at the surface is important for sustained operation. The relative length of loading and resting phases is one of the most important operational parameters. During the initial years, the loading phase may be relatively short, maybe just a couple of days, followed by a two to six week resting period. Once the beds gets older and the layer of accumulated sludge in the beds thicker, the loading period may be longer (maybe one week) followed by an equivalent longer resting period. The maximum drainage velocity during loading is a good indicator that can be used to decide on the duration of the loading period.

Sludge quality

The majority of planted reed beds in Denmark are designed to process sludge from urban domestic wastewater treatment systems using activated sludge as the treatment process. As treatment demands in Denmark include efficient removal of both nitrogen and phosphorus, these systems usually operate with relatively long detention times and a high sludge age to secure efficient nitrification and denitrification. Phosphorus is usually removed by biological P-removal supplemented with chemical precipitation by iron and/or aluminium salts. Because of the long detention times of the sludge in the activated sludge systems, the sludge delivered to the planted reed beds are usually ‘stabilised’ and have a low oxygen demand.

The sludge quality varies significantly between systems as well as over time in the individual systems. The sludge quality is, however, very important for the ability of planted reed beds to dewater and mineralise the sludge. In general, good results have been obtained for ‘normal’ stabilised activated sludge from urban domestic wastewater treatment systems, and systems have performed well at a loading rate of 60 kg dry matter per square metre per year. However, some systems have experienced problems that can be associated with sludge quality. The knowledge about sludge quality parameters and how these affect capacity and performance of planted reed beds is still insufficient.

Effective biomass–water separation is essential to the activated sludge process in biological wastewater treatment. Bioflocculation transforms bacterial cells into sludge flocs, which facilitates biosolids–water separation including sludge sedimentation, compression, and dewatering. Extracellular polymeric substances (EPS, biopolymers that surround the bacterial cells) are believed to play an important role in the formation of sludge flocs. However, the exact effects of EPS on bioflocculation and sludge–water separation are still unclear. The EPS abundance of the sludge can be sensitive to the environmental variations, such as changes in organic loading rate, detention time, and carbon and nitrogen sources. It is likely, that any factor in the wastewater treatment plant that will influence the amount of EPS is also likely to influence the dewatering in the planted reed beds.

The following parameters have all been shown – or suggested – to be associated with operational problems in planted reed beds:

- Mixing of sludge from an anaerobic digester with activated sludge prior to dewatering in the planted reed beds. This may result in a mixed sludge with a higher oxygen demand that has different dewatering properties
- The physical influence of using certain types of high-pressure pumps for transferring the activated sludge to the planted reed beds may disrupt floc structure and negatively impact dewatering
- Transport of sludge in long transport pipes and/or storage of sludge prior to spreading on the surface of the reed beds have been shown to negatively affect sludge dewatering in the reed beds.
- Sludge from wastewater treatment systems treating a significant fraction of industrial (including agro-industrial) wastewater have been shown to be more difficult to dewater. Some studies indicate that the content fat in the sludge may be correlated with the dewatering difficulties.
- Addition of precipitation chemicals for phosphorus removal may influence dewatering properties.

Because of the coherence between sludge quality and dewatering properties, it is very important in any project, prior to dimensioning, to assess the sludge quality, including its degree of stabilisation, e.g. its oxygen demand, its bio-flocculation properties and its dewatering characteristics. It has been suggested to use simple laboratory measurements such as ‘Specific Resistance to Filtration’ and ‘Capillary Suction Times’ as means to assess dewatering properties of the sludge, but the relation to sludge dewatering in planted reed beds has not as yet been established.

DISCUSSION

Processing of surplus sludge from wastewater treatment systems can be efficiently carried out in planted reed beds. This approach has several advantages compared to the conventionally used handling techniques. In addition to dewatering, the organic matter in the sludge is partly mineralised, thereby minimising the sludge volume and improving the sludge quality. The overall sludge volume reduction occurs without the use of chemicals and involves only a very low level of energy consumption for pumping sludge and reject water. The final sludge product can safely be applied as a fertilizer on agricultural lands.

Although planted reed bed systems in theory are relatively simple systems, that rely largely on natural processes, experience has shown, that it is very important to understand and respect the basic design and operation requirements. Experience from Denmark has proven that if some of these requirements are not respected, the systems are likely to fail. Still, there are a number of outstanding issues that are not fully understood, particularly in relation to the effect of sludge quality. There are no accepted means to evaluate the ease with which a sludge will release its water, and even less so for evaluating how a sludge will be dewatered in a planted reed bed. Therefore, extreme care should be taken when attempting to extrapolate the use of planted reed beds to applications and regions outside of its ‘normal’ and documented area of application. Having said this, there is however no doubt that planted reed beds have a great application potential in other climatic regions and for several types of sludge. Applications in subtropical and tropical areas are still on an experimental scale. However, planted reed beds are likely to perform more efficiently in warm and dry climates due to the more benign and stable temperatures, which should accelerate the rate of biological processes.

Sludge Treatment in Reed Beds Systems: Development and experience through 22 years in Denmark

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Abstract
There are important differences in the environmental perspectives and costs involved in mechanical sludge dewatering followed by disposal on agricultural land compared to reed bed sludge treatment. The background for the evaluation of their influence on the environment is to a large extent the experience gained since 1988 from the operation of the large number of reed beds systems currently operating in Denmark and Europe. The effect on the environment of the establishment and operation of a Sludge Treatment in Reed Beds system (STRB) is seen as quite limited compared to traditional sludge treatment systems such as mechanical dewatering and drying, with their accompanying use of chemicals; incineration; direct deposition on landfill sites, etc. After reduction, dewatering, and mineralisation in a reed bed sludge treatment system, sludge with a solids content of 0.5-3% can attain dry solids content of up to 20-40%. In addition, mineralisation removes up to 25% of the organic matter in sludge. Experience has shown that the quality of the final product in sludge reed beds with respect to pathogen removal and mineralisation of hazardous organic compounds after treatment make it possible to recycle the biosolids to agriculture.

Keywords
Sludge dewatering, loading, reed beds, emptying, sludge quality, environmental impact.

LOADING – OPERATIONAL STRATEGY

The operation of a reed bed system may be divided into a number of periods relating to the lifetime of the system. A system generally runs for a total of at least 30 years; this period is divided into two or three 8-12 year phases. Each phase consists of commissioning, normal loading, emptying and re-establishment of the system. Full operation following the commissioning of the plant operations means that the yearly loading is increased to the sludge production from the wastewater treatment plant corresponding to the maximum capacity (tons dry solid/year) of the sludge reed bed system. The loading strategy involves assigning an individual quota to each individual basin. This quota is a sludge volume which generally increases throughout the entire period of operation until emptying, but it may also vary or even decrease to zero for periods. The length of the loading periods and rest periods between loadings depends on the age of the system/basin, the dry solid content, the thickness of the sludge residue and the intensity of partial loadings during the period of loading. On a daily basis, the basins are subjected to a loading of 1-3 partial loadings of approx. 1 hour for a short period (from a few days to a maximum of 2 weeks during commissioning) until the quota is used and loading switches to the next basin.

Mechanical sludge dewatering involves conditioning with chemicals, usually in connection with the dewatering process itself. Either organic polyelectrolytes or inorganic conditioning substances are used (Table 1). In a sludge reed bed system the dewatering process is governed by the sludge quality, the climate, the wind, the gravity and the vegetation. The water in sludge with a dry solid content of 5% can be divided into pore water (66.7%), capillary water (25%), adsorption water and structurally bound water (8.3%). Dewatering the pore water concentrates the sludge to a dry solid (DS) content of about 15%. Further dewatering by removal of the capillary water concentrates the sludge up to a dry solid content of about 50%. The remainder of the water in the sludge may be removed by drying. Reed beds have been used for sludge reduction in Denmark and Europe since 1988 when the first sludge processing system was introduced. Long-term sludge reduction takes place in reed-planted basins, partly due to dewatering (draining, evapotranspiration) and partly due to mineralisation of the organic matter in the sludge. From waste-water treatment plants the sludge is pumped onto the basin surface/sludge residue. The dewatering phase thus results in the dry solid content of the sludge remaining on the basin surface as sludge residue, whereas the majority of its water content continues to flow vertically through the sludge residue and filter layer. The sludge residue water content is further reduced through evapotranspiration. In addition to dewatering, the organic matter in the sludge is mineralised, thereby minimising the sludge volume. The overall sludge volume reduction occurs without the use of chemicals and involves only a very low level of energy consumption for pumping sludge and reject water. Experience from the reference plants is that this type of system is capable of treating many types of sludge with a dry solid of approx. 0.5 to approx. 3-5%.

Table 1. Dry solid content of treated sludge related to dewatering method

Dewatering method	Centrifuge	Filter Belt Press	Filter Press	Traditional Sludge Bed	Sludge Reed Beds Systems
% Dry Solid	23 (15-20*)	24 (15-20*)	32	10 **	20 – 40

* Normally observed values. ** Variable, depending upon the duration of the treatment period.

Periods of operation

The system runs at full capacity for subsequent 10-year periods of operation, including periods of emptying. Normally, emptying is planned to start in year 8 and is completed in year 12 of each operation period. In order to meet the requirements of capacity for a 10-year treatment period of operation, as well as dewatering of the sludge residue to a dry matter content of approx. 30 - 40% (Table 1), the following dimensioning standards are recommended. Dimensioning of the sludge reed bed systems is based on the following factors: Sludge production (tons of dry solid per year), sludge quality, sludge type and climate.

Sludge quality

The physical quality of the sludge changes at different stages of the dewatering process. The content of fat (max 5,000 mg/kg DS) in the sludge, as well as the form of production (e.g. low sludge age, concentration, pre-dewatering using polymer, mesophile or thermophile digestion) are of importance to the sludge dewatering capacity and to the final dimensioning and number of basins. In addition to the sludge dewatering capacity, loss on ignition is a factor in the dimensioning. As a rule, a loss on ignition of 50-65% is recommended.

Areal loading rate

The areal loading rate is determined in relation to the sludge type, climate and must take emptying into account. With regard to loading of surplus activated sludge, the areal loading rate is set to maximum 30 - 60 kg DS/m²/year after commissioning. With regard to sludge types, e.g. from digesters (mesophile, thermophile), sludge with a high fat content, or sludge with a low sludge age (< 20 days), an areal loading rate of maximum 30 - 50 kg DS/m²/year is recommended.

Number of basins

In relation to a 10-year period of operation, dewatering, vegetation and mineralisation, it is necessary to operate the basins with alternating periods of loading and resting. Regardless of sludge type and the size of sludge production, a minimum of 8 – 10 basins are necessary, in order to achieve the required ratio between loading and resting periods. Experience shows that systems with too few basins, i.e. fewer than 8, often run into operating problems, including very short periods of operation until emptying with poorly dewatered sludge residues and poor mineralisation. The basin depth must be no less than 1.70-1.80 m from the filter surface to the crown edge. The basins must have a sufficiently high freeboard to allow for 1.50-1.60 m of sludge residue accumulation. Basin capacity must also allow for increased loadings during the emptying phase of e.g. 2-4 years until all basins have been emptied.

Figure 1. Kolding Sludge Reed Bed System (September 2000).

SYSTEM DESCRIPTION AND DESIGN

Sludge from the wastewater treatment plant the sludge may be pumped out from the active sludge plant, final settling tanks, concentration tanks or digesters in batches into the basins.

Filter design and reeds

Each basin forms a unit consisting of a membrane, filter, sludge loading system and reject water and aeration system. (Fig. 1). The total filter height is approx. 0.55-0.60 m before sludge loading. The reeds contribute to dewatering the sludge via increased evapotranspiration from the sludge residue and by mechanically influencing the sludge residue and filter. Finally, the presence of reeds contributes to the mineralisation of the organic solid in the sludge.

Sludge loading, reject water and aeration systems

Loading must be planned in such a way as not to inhibit development of the reeds and to prevent the sludge residue from growing so fast that the reeds cannot keep up horizontally and vertically. It is not recommended to apply a 100% loading rate immediately after planting. Pressure pipes are installed to each basin, terminating in a distribution system to distribute the sludge. An important detail is to ensure that the sludge is pumped out in a way which creates a uniform and even sludge load across the entire filter area. The reject water system has two functions. The first is to collect and return the filtered water to the wastewater treatment plant. The second function is to aerate the filter and the sludge residue.

ENVIRONMENTAL IMPACT ASSESSMENT

Sludge treatment in reed bed systems is a thoroughly tested method with a number of proven advantages. Experience shows that the method is an environmentally friendly and cost efficient sludge treatment. It uses very little energy and no chemicals, has a minimum of CO_2-emissions, provides a good working environment, and reduces sludge residue significantly. The European Union framework directive calling for cleaner discharges from our waste water treatment facilities can result in more sludge, due to the improved treatment; however, managing sludge is rather costly. In countries like Denmark, Germany, France and Sweden sludge treatment in reed bed systems are a common and a well-proven method during the last 22 years.

Better working environment
When the system is setup, there is no contact with the sludge. There is no noise from the system as there is from many other types of treatment systems, and there is no odour from it either. The system works effectively to reduce pathogenic bacteria like *Salmonella*, Enterococci and *E. coli*, thus making it a lot safer to be on site.

A cost effective system
The man-hours needed to run the system are fewer than with traditional methods, and require only a weekly control-visit to the site of about one to two hours. Sludge treatment reed bed systems utilise the forces of nature to reduce and treat sludge. The only appreciable power consumption is by the pumps used to transport sludge and reject water. This means that the reed bed system uses much less power than other systems. Transport costs will be reduced substantially, while the volume of sludge can be reduced to approximately 1.5-2.5 % of its original volume. The sludge will be of a better quality and suited for use on agricultural land. This offers more opportunities for disposing of the sludge after treatment.

N**o chemicals needed**
Sludge treatment in reed bed systems uses no chemicals in the dewatering process. This means a considerable improvement in the working environment along with a reduction of the chemical residue in the treated waste water passing into the environment.

Good options for recycling
The content of substances in sludge that are foreign to the environment can be reduced to such a degree that the sludge conforms to the limits and norms for deposition on agricultural land. Treatment in a sludge reed bed system was shown to be effective at treating raw sludge containing large amounts of pathogenic bacteria including *Salmonella*, Enterococci and *E. Coli*. As a general rule, pathogenic bacteria that are excreted and end in an alien environment only live for a short period of time, depending upon various environmental factors and the bacteria's own characteristics.

The sludge (approximately 0.5-0.8% DS) loaded into the individual basins contained a large number of bacteria. *Salmonella*, Enterococci and *E. Coli* were found in the sludge in the following quantities: 10-300 per 100g (wet weight), 7,000 – 25,0000 CFU/g (wet weight) and 800,000 – 10,000,000 CFU/100g (wet weight), respectively. Analysis of the reduction in pathogens in the sludge residue through a period of 1-4 months after the last loading from the Helsinge sludge reed bed system (basin no. 8) indicated that the pathogen content was reduced to <2 per 100g (*Salmonella*), <10 CFU/g (Enterococci) and <200 number/100g (*E. Coli*). For Enterococci and *E. Coli* the reduction was approximately 5 log units and 6-7 log units, respectively.

Mineralisation of Linear Alkylbenzene Sulphonates (LAS) and Nonylphenolethoxylates (NPE), which may be detrimental to the environment, if spread in large concentrations, in mesophilic digested sludge was observed during a 9 month monitoring programme. The reduction was 98% for LAS and 93% NPE. After the treatment of sludge, recycling options are good, particularly in agriculture. The sludge quality is cleaner and more adaptable in the natural cycle than mechanically dewatered sludge.

CASE STUDIES
Experience from a large number of systems in many countries treating a whole range of sludge types has shown the efficiency of the method, which can be demonstrated in selected cases.

Helsinge sludge reed bed system

Sludge production from the Helsinge wastewater treatment plant consists of activated sludge directly from the activated sludge plant and activated sludge from final settling tanks. This production (tons dry solids) constitutes approximately 66% of the loading of the sludge reed bed system. The remaining 33% of the sludge production consists of concentrated anaerobic activated sludge from 4 smaller wastewater treatment plants. The type of sludge is mixed in each delivery before being added to the reed bed system. The sludge is pumped via a mixing tank and a valve building, where the sludge flow and dry solids are registered before being led to the respective basins. Total sludge production has increased during the period from 1997-2005 from approx. 209 tons of dry solid annually to approx. 606 tons of dry solid annually. Annual sludge production amounts to 550 - 600 TDS (Fig. 2).

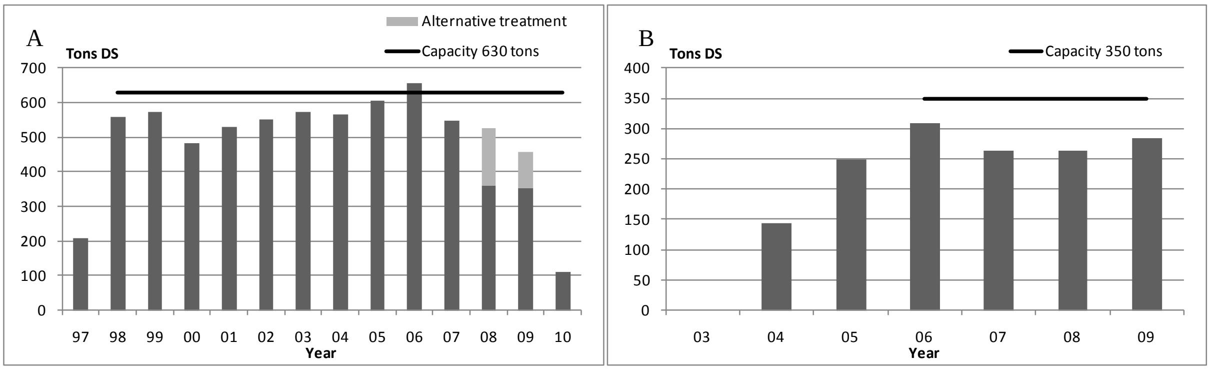

Figure 2. Sludge production and sludge load. A. Helsinge, B. Nordborg.

The Helsinge sludge reed bed system was established in 1996 and consists of 10 basins, each having an area of 1,050 m^2 at the filter surface. The system has a capacity of 630 TDS per year and a maximum area loading rate of 60 kg DS/m^2/year. The annual load rate (tons dry solids) of the Helsinge sludge reed bed system during the period from start of operations and to 2010 has been in the order of 90 % of capacity. From 2000 to 2005, loading has increased by approx. 130 tons dry solids. The loading regime of the system consists of applications of approximately 130-150 m^3 of sludge (mixed sludge) being applied once or twice daily to the plant's basins in relation to individual basins' loading quota and capacity, with the feed concentration being approximately 0.5-0.8 % DS. Each basin was subjected to a loading quota of 1,500 m^3 over a period of approximately 6-8 days. Loading was followed by 45-65 rest days.

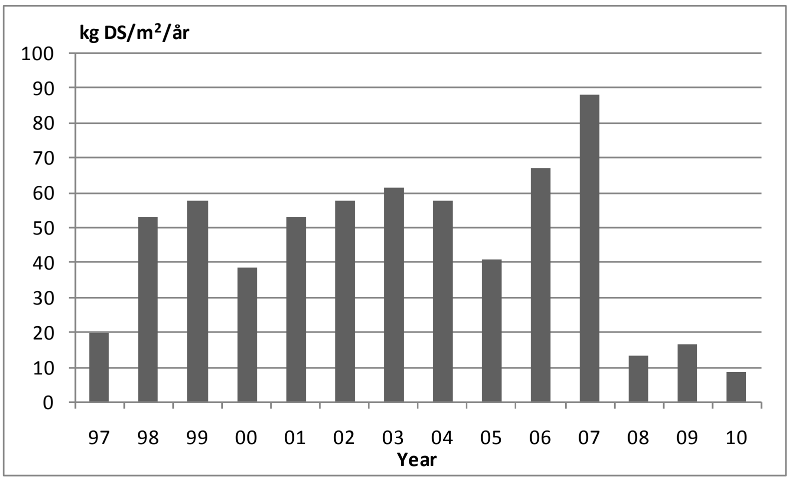

Figure 3. Helsinge sludge reed bed system (basin no. 1) – Average area loading rate (kg dry solid/m^2/year).

After commissioning the individual basins were subjected to an average loading rate of approximately 55-65 TDS per year, resulting in an average area-specific loading rate of 55-64kg DS/m^2/year. Because of the increasing sludge production and emptying of two basins yearly, the area-specific loading rate has increased from approximately 46 kg DS/m^2/year in 2000 to 68-88 kg DS/m^2/year in 2007. The sludge residue height status in basin 1 in relation to time and area-specific loading rate (kg DS/m^2/year) was calculated on the basis of scale pole readings. The sludge residue height increase from 1998 to 2005 was approx. 1.20 m (basin no. 1), and the total sludge residue height by April 2008 was approximately 1.60 m (Fig. 4). Helsinge sludge reed bed system has been emptied over a 4-year period (2005-2008), with 2-3 out of 10 basins selected for emptying per year. Capacity during the emptying period was maintained at 630 tons of dry solids per year. The last 3 basins were emptied in 2008.

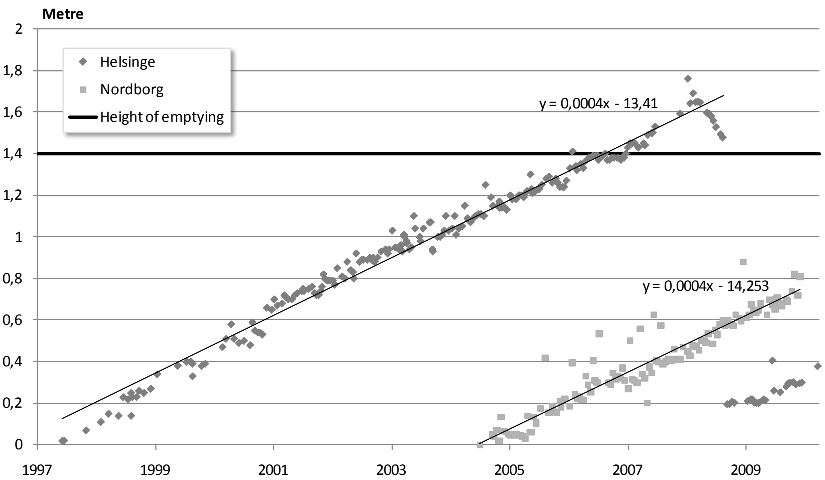

Figure 4. Sludge residue increment.

Sludge residue quality

The quality of the sludge residue in the Helsinge sludge reed bed system met valid statutory order criteria with regard to heavy metals and hazardous organic compounds for use on agricultural land after ten years of biological treatment in the sludge reed bed system. The dry solids content in the sludge residue was up to 35.5%. Nitrogen and Phosphorus contents were on the order of 22,000-28,000 and 30,000 mg/kg DS, respectively.

Emptying, Recycling and Regeneration

The plan was to empty the Helsinge over a 5-year period (2005-2008) with 2-3 out of 10 basins selected for emptying per year. The two basins selected for emptying are excluded from the loading plan approximately ½-1 year before emptying, and have a reduced load the first year after emptying. Capacity during the emptying period (5 years) was maintained at 630 tons of dry solids per year despite reduction of the basin number during the emptying period. The loading of individual basins increased from approximately 55 to 88 kg DS/m^2/year, as system loading during the emptying period applies to only 6-8 basins. From each of the basins approximately 1,000-1,400 tons of sludge residue were removed. The sludge residue was deposited on approx. 158-170 ha taking into consideration Phosphorus content with max. 90 kg P/ha for individual areas every third year. Maintaining full capacity during emptying is only possible provided that the basins are re-established after emptying with sufficient regeneration of vegetation, and provided that the loading rate is adapted to vegetation growth. Helsinge sludge reed bed system has generally had a satisfactory rate of regeneration after emptying in both 2005 and 2006, so that re-planting basins has only been necessary in few of the 10 basins.

Nordborg sludge reed bed system

Nordborg sludge reed bed system is another Danish system with 10 basins. The system was established with reeds in 2003 and has a capacity of 350 tons of DS per year. Each of the basins has an area of approx. 705 m^2 at the filter surface and a maximum area-loading rate of 50 kg DS/m^2/year. Sludge production from the WWTP consists of activated sludge (SAS) directly from the activated sludge plant and digested sludge from a mesophilic digester. The two sludge types are mixed before being added to the reed bed system. 90 – 120 m^3 (approx. 0.5 % DS.) of SAS is mixed with 3-6 m^3 of digested sludge (approx. 2-3 % DS). The Annual sludge production amounts to 250 - 300 ton of DS (Fig. 2).

Finally, the batch is diluted with effluent from the WWTP to a final volume of 140-160 m^3. The system's loading regime consists of applications of approximately 140-160 m^3 of sludge (approx. 0.6-08 % DS) once daily. From 2006, each basin was subjected to a loading quota of 600 m^3 over a period of approximately 4 days. Loading was followed by a 36-64 days rest period. The area-specific loading rate was between 36-44 kg DS/m^2/year in period 2004 to 2009. The sludge residue height increased in the period from 2004 to 2009 by 0.83 m (Fig. 4). The plan is to empty Nordborg sludge reed bed system over a 4-year period (2011-2014), with 2-3 out of 10 basins selected for emptying per year. Capacity during the emptying period will be maintained at 350 tons of dry solids per year.

Figure 5. Nordborg Sludge Reed Bed System

Hanningfield sludge reed bed system

The Hanningfield (England) sludge reed bed system is a new system treating water works sludge. The use of reed bed systems not only reduces the capital and operating cost, but also provides the site with an environmentally-friendly operational area. Therefore, 6 trials bed (20 m^2 each) have been monitored (2008 – 2010) to examine the dewatering processes of the liquid sludge produced from the water treatment process, which includes treatment with iron sulphate to help dirt particles to coagulate. It is possible to get the vegetation to grow in ferric sludge, where the pH was measured to 7, 7. It has not been necessary to use fertilizer. The influence of the loading programs (15-50 kg DS/m^2/year) was tested. It is possible to drain and treat ferric sludge (approximately 300,000 mg Fe/kg DS). Generally the dewatering profile is a peak with a maximum over 0.010 – 0.025 l/sec/m^2. The times for dewatering of 6-12 m^3 are approximately 15 hours and over 90 % of the load is dewatered in that period. Even for the basins which had been loaded with 12 m^3 each day for 4-6 days. The dry solid (0.13 - 0.20 %) in the sludge has been concentrated approximately 200 times. The dewatering phase results in ferric sludge with 30-40% dry solid which cracks up very quickly. In spite of the different loading programs, volume reduction is very high at over 99 %.

Figure 6. Hanningfield Sludge Reed Bed Systems (June 2009)

CONCLUSIONS

This paper presents experience and know-how from a 22-year period (1988-2010), primarily with references from Denmark. The accumulation of knowledge, guidelines for dimensioning and operations, and descriptions are based on experience from more than 10 sludge reed bed systems, mainly loaded with activated sludge residue. The sludge treatment reed bed system is a long-term sludge solution, and the systems are built to treat sludge for an average operative period of 10 years. Experience shows that operation is reliable and flexible, with very low operating costs, low energy consumption, no use of chemicals (polymers) for dewatering, an improved working environment and the freeing-up of waste water treatment capacity. The basins in Helsinge sludge reed bed system have, since 1998, been subjected to an average loading rate of approx. 55 tons dry solid per year, resulting in an average area-specific loading rate of 55-64 kg DS/m^2/year. The sludge residue height in relation to time and area-specific loading rate increased from 1998 to 2008 was approximately 1.60 m, or approximately 0.16 m per year. It has been possible to maintain full capacity during emptying because the basins are re-established after emptying with sufficient regeneration of vegetation.

It has thus not been necessary to re-plant basins. Experience shows that good mineralisation of hazardous organic compounds, good reduction of pathogenic microorganisms and a final dry solids content of 30 - 40% can be achieved. With respect to heavy metals, hazardous organic compounds and pathogen removal 10 years of treatment make it possible to recycle the biosolids on agricultural land.

Sludge treatment by drying reed beds: The French experience.

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Abstract
Sludge drying reed beds exist in France since the early 1990s. These facilities often consisting of a minimum of 6 beds each being alternately fed during a week. Some of the firsts sludge drying reed beds have known deficient performances in term of final dry matter content. The cause of these dysfunctions are generally attributed to a poor number of units. New researches started in 2005 to precise design rules and operating strategies for activated sludge treatment and septage as well. This paper resume briefly the new design and feeding strategies to obtain better performances. These experiments draw positive conclusion in term especially for septage treatment.

Keywords
Sludge drying reed beds, activated sludge, septage, design, operating strategies

HISTORICAL DEVELOPMENT IN FRANCE

First experiments of sludge drying reed beds [SDRB] began in the early 1990s in France (Liénard et al., 1990, 1995) under Cemagref researches at the request of the SAUR company. Done on 20 m² pilots scale, this experiments allowed to draw the firsts design recommendations. The process has really been in use since 1996 (Fig. 1). In 2003, about 163 systems had been built in France with a treatment capacity between 200-12000 people equivalent [p.e.]. Main plants are developed for small communities (700 to 4000 p.e.) while nowadays there is a tendency to increase the size capacity. In 2010, there are approximately 300 facilities and the biggest one is sized for 27000 p.e. (Fig. 1).

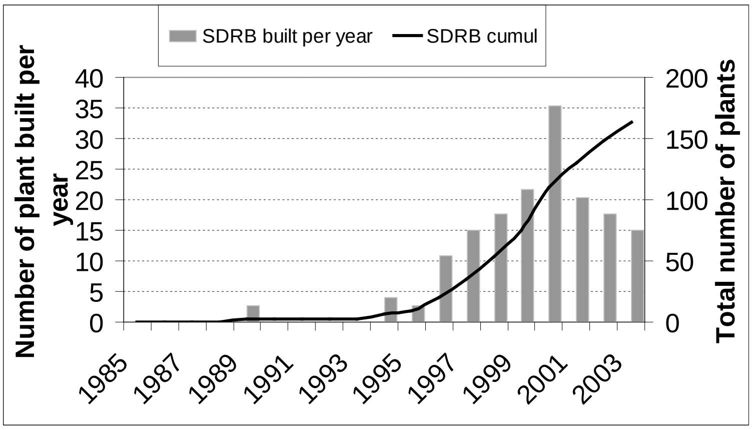

Figure 1. Number of sludge drying reed beds in France drying beds planted with macrophytes in operation (September 2003)

Figure 2. 2 m² pilots of the experimental Andancette plant.

The first guidelines recommended a minimum of 4 parallel beds on a basis of 50 kg of dry matter per m² and per year, the sludge being directly taken in the aerated tank to control perfectly the organic load. Feedback from the field (Lesavre et al., 2002) has shown that many companies built plant with less than 4 beds in parallel (especially for small size capacity) and the dry matter content [DM] of the sludge, when withdrawing, were only about 10-15 %; whereas in Denmark, (Nielsen, 2003; 2005) observed a DM content up to 30-40% with 8 or more units. Indeed we observed that only 15 % of dry matter content can be ensured with 4 units (Liénard, 1999). The lack of knowledge of the concerned mechanisms tend to empiricism and uncertainties in bed design (number of beds, composition of the filtration layer, passive aeration design,…) and operational strategy (organic load, hydraulic load, feeding/rest periods,…) which can lead to anaerobic conditions and poor vegetation growth (essentially during the commissioning phase), insufficient drainage and clogging phenomenon (Nielsen, 2003; 2005). From this, Cemagref started a new research program in 2006, in collaboration with SINT and Véolia companies and the French water agency. The aims were multiple :

- To optimize activated sludge treatment by SDRB (bed's conception, number of units, loads…)
- To transpose SDRB systems to septage treatment
- To determine later if more points

The experiments were performed on 16 experimental concrete beds of 2m2 each built close to an extended aeration activated sludge plant (Andancette, 13 000 p.e., France) (Troesch et al., 2009a, 2009b, Vincent et al., 2010) (Fig. 2).
This paper resume the actual guideline and knowledge of the French experience according to the application fields (activated sludge, septage, primary sludge).

DESIGN AND OPERATION RECOMMANDATIONS FOR ACTIVATED SLUDGE

Global sizing of the beds is based on the sludge production and the organic load in kg of suspended solids [SS] per m² and per year. Initially the organic load was expressed in term of dry matter. We changed it when working on specific sludge with high content of dissolved salts (septage) which do not impact the clogging of the bed.

Sludge production

Sludge production is the first step of the sizing of the beds. It can be estimated on the basis of :

$$DM = C\left(\frac{BOD + SS}{2}\right)$$

Where :

DM, BOD and SS are expressed in $kg.d^{-1}$

C is a coefficient equal to 0.84 in the case of separated sewer and up to 1.02 in the case of combined sewer.

In case of physico-chemical removal of phosphorus, the sludge production has to be increased. Actually, In France, on the basis of a daily production of 2.5 g $p.e.^{-1}$, the sludge production is increased of about 10 %.

Organic load and number of units

The functioning of SDRB systems depends on the maintaining of aerobic conditions to favour sludge mineralization, reed growth and, by the same way, the drainage and the drying of the sludge. As a consequence, the organic load applied on the beds has to be controlled to avoid the appearance of anaerobic conditions.

The total annual load must to never overpass 60 kg $SS.m^{-2}.y^{-1}$. As some units have to be stopped when planning the emptying of the sludge, the unit surface has to be calculated to not overpass the maximal load on the other units still working in emptying operation. According to the number of units, the Table 1 presents the loads applied during the emptying operation and during normal functioning of the units.

Table 1. Impact of a 5 months emptying phase on the surface loads applied on the units.

Total Number of units	Total length of a cycle before emptying	Surface load on all the units kg $SS.m^{-2}.y^{-1}$	Surface load on the units during emptying phase kg $SS.m^{-2}.y^{-1}$
6	3 years	48	60.7
8	4 years	50	60.1
10	5 years	50	56

The final dry matter content of the sludge is greatly impacted by the load and the number of units. It is not possible to exceed 15% of dry matter content when using only 4 units. When sizing RBDS for communities of more than 3000 p.e., we recommend a minimum of 8 units to allow a long enough rest period before emptying. This configuration allow to guarantee 30 % of dry matter content. Nevertheless, for small communities (lower than 2000 p.e.) it can be economically more interesting to realize 4 units designed with a total surface load of 30 kg $SS.m^{-2}.y^{-1}$. In that configuration 30 % of dry matter can also be obtained.
The performances of the system greatly depend on the reeds development. We estimate that, during the commissioning period, the organic load has to be maintained lower than 25 to 30 kg $SS.m^{-2}.y^{-1}$ for that. This period lasts until reaching a stems density of 250 per m².

Beds conception
The bed's conception is presented in the Fig 3. It is composed, from bottom to the top, of :

– A pebble layer of 15 cm min. in which drainage/aeration pipes are positioned.
– A transition layer of 10 cm or a three-dimensional geogrid to avoid gravel to migrate into the pebbles.
– A gravel layer with particle size strictly larger than 2-4 mm. Use 3-6 mm appears to be a good compromise.
– A filtration layer of vegetal compost from a composting platform according to the French standard NF U 44-051.

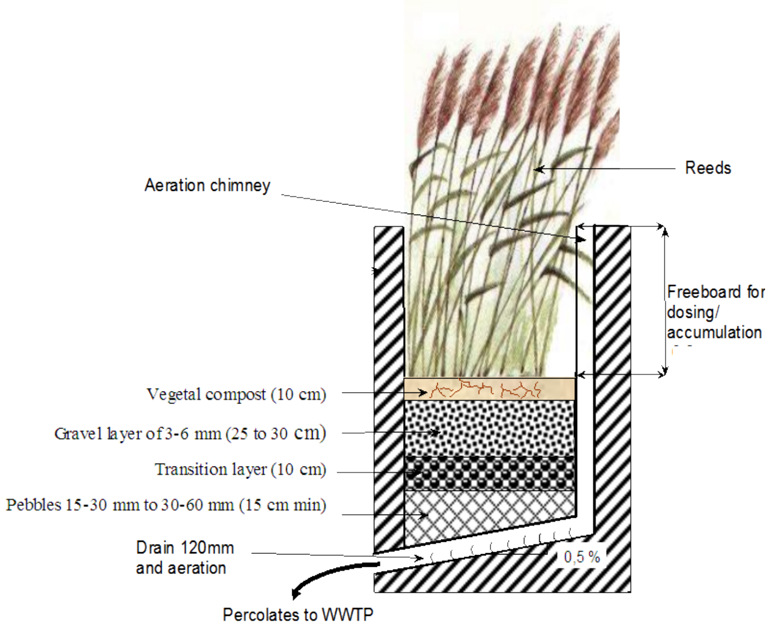

Figure 3. Cross section of a unit

Contrary to old French recommendations (in which the filtration layer was composed of sand, Liénard et al., 1995), it is now recommended to use vegetal compost. This choice is based on the fact that the vegetal compost ensures a better growing media (reeds growths faster) and favours sludge infiltration rate, thus media oxygenation, probably due to a better capillary connection with the residual sludge compared to sand. (Troesch et al., 2009a).

The passive aeration system consists of drain pipes, placed every 3 m, which collect the leachate in the bottom layer and are connected to the atmosphere on opposite sides. The units are fed directly from the aeration tank (homogeneous product representative of the sludge level of the plant) once or twice a day at a flow rate greater than 0.25 $m^3.h^{-1}$ per m² of unit. To optimize distribution of the sludge onto the unit surface, a minimum of one point distribution per 100 m² is needed. It appears that using sedimentation tank to fed the beds with thickened aerated sludge it is not useful. Experiments done by Troesch et al. (2009a) showed that it is detrimental for beds conditions (lower redox potential) and final sludge quality.

Operation mode

The high organic load applied onto the beds and the organic (70 – 80 % of volatile solids) and fermentable aspect of the sludge can lead to anaerobic conditions when loads are excessive or when the operation mode is not meticulous, especially in winter. Thus operation mode has to be taken seriously by several rules and common sense. The objectives of the operation mode (frequency of feeding/resting periods) are:

- To ensure an optimal reed growth
- To promote aerobic condition to mineralize the accumulated organic matter
- To dry the sludge by evapotranspiration

On one hand, long rest period will favor aerobic conditions and the drying of the sludge but, on the other hand, such operation can lead to reeds water stress. As the accumulated sludge serves as a water stock the feeding/rest period has to be adapted to the sludge accumulation, i.e. the years of functioning since the last withdrawal. The Table 2, adapted from Nielsen (2003), presents the French recommendations on this point.

Table 2. Feeding/rest period strategies according to the time functioning since the last withdrawal.

Year of functioning	Feeding period (day)	Rest period (day)
1	2 – 3	10-20
2	4 – 5	25 – 40
3	5 – 6	30 – 55
4	6 – 7	35 – 65
8	7 – 14	70 – 120

In the French climatic context, two periods can be critical. First of all, reeds can suffer water stress in summer during the first year of functioning especially since the beds work at half load. In such condition the owner has to give heed and accelerate the rotation between feeding and rest period. In some specific cases, saturation of the media (maintaining the water level below the beds surface) can be done during 2 to 3 weeks max. in rest period, to promote reed development. It has to be emptied before the new feeding phase.

The other critical period can occur in spring when too long feeding period is practiced (14 days for example) in winter season while the units receive their nominal load. The organic matter accumulated during winter can lead to oxygenation deficiency when bacteria com back to an intense activity with the increase of temperature. Such condition can be detrimental to reeds growth and, consequently, to drying efficiency. It is thus recommended to not over pass 7 days of feeding period in winter season.

As aeration of the media is an essential point guaranteeing the functioning of the system, it is important to supervise the hydraulic efficiency of the units. Poor infiltration rate, due to clogging, will diminish oxygen diffusion into the media. Consequently, to follow the peak flow after a batch feeding is a simple and essential way to obtain data useful in operation strategy. Steen Nielsen suggests to change the unit to be fed when the peak flow is smaller than 0.3 $L.min^{-1}.m^{-2}$. As the hydraulic of the system depends, on the one hand, on the media characteristics and, on the other hand, on the feeding flow, it is delicate to transpose Danish recommendation to the French conception of the beds. We generally observe greater peak flows than the limit mentioned above. However this may be, measuring the drainage flow is important to do to give the owner some minimum tools he can use to improve the units management.

Performances

Percolates quality

Using vegetal compost in spite of sand as the upper filtration layer induce a decrease in filtration efficiency during the commissioning period. Nevertheless, SS removal still greater than 96 % with 10 cm of vegetal compost. Once the sludge layer increases the filtration efficiency is enhanced. The Table 3 presents the percolates quality during the commissioning period and normal operating period. Once sludge layer is present filtration efficiency becomes effective for all particles greater than 3 µm.

Table 3. Percolates quality during commissioning period and normal operating mode period when using vegetal compost as the upper filtration layer.

	Commissioning period		Normal operating mode	
	mean	max	mean	max
SS ($mg \cdot L^{-1}$)	70	610	10	40
COD ($mg \cdot L^{-1}$)	105	510	45	70
KN-N ($mg \cdot L^{-1}$)	7	33	4	16
NH4-N ($mg \cdot L^{-1}$)	2	10	2	13

Maximum values appear the first day of feeding when sludge is crackled due to the drying during rest period. The pollutant fluxes of percolates coming back to the inlet of the WWTP represents about 1 % for COD and KN, and about 8,5 % of the Total Phosphorus (assuming no physico-chemical treatment of phosphorus.)

Sludge quality

The final sludge DM content (during the withdrawal) depends on the maintained DM content during the process. 24 h after a batch feeding, about 90 % of the water is drained : only 8 ± 4 % of DM content can be reach by this way. Thus the rest period is of importance to dry correctly the sludge by evapotranspiration. We observed that if we want to reach more than 25 % of DM content at the final, the owner has to maintain a minimum of 17 % of DM content at the end of each rest period. It is not problematic in summer season in France, but has to be looked out during winter, and adapt the operating mode if necessary. By this way, the final sludge DM content is higher than 25 – 30 % with about 60 % of VS.

Due to rhizosphere effect, this residual sludge is comparable to a low deformable and well-structured material easy to spread on agricultural land (Troesch et al., 2009a). The status of solid and stabilized sludge could be claimed for this sludge and, according to the French regulation, it can be extracted from the reed beds and temporarily stored on plots near the fields where it will be further spread if the harvest is not already done. The dissociation of emptying and spreading should give more flexibility to respect the ideal period to empty the bed(s) which must be scheduled between July 15th and late August to benefit both of the high evapotranspiration in summer and not to undermine the regrowth of reeds in the beds emptied before winter.

SEPTAGE TREATMENT

We know actually a expansion of the application fields of SDRB. Used sometimes for primary sludge (Torrens et al., 2006), there is a growing interest in developing such systems for septage. This is related to the obligation of regulatory control of onsite wastewater treatment of dwelling houses, which increases the volumes of septage. Its main destination is direct agricultural reuse or co-treatment with wastewater in treatment plants larger than 10,000 P.E. While the first solution is not well accepted (sanitary risks, high septicity and ammonia concentration leading to odour inconveniences), the second is not always achievable. In fact large wastewater treatment plants are either not so numerous in rural areas (transport cost) or not systematically able to treat an additional organic load. As septage is the result of anaerobic processes in the septic tank, it is composed of a lot of tiny and non-flocculated particles in suspension in a liquid fraction where many salts are dissolved (ammoniacal nitrogen, volatile fatty acids, orthophosphates, hydrogen carbonates…) resulting in high electric conductivity values (2,500 $mS \cdot cm^{-1}$). The main consequences are, a difficult solid/liquid separation and a decrease of the dewaterability. Therefore different sludge preparations have been tested in France like:

- Treating directly septage on the reed beds: the interest results in the simplicity and the economical aspect of such a process.
- Mixing septage with aerated sludge: the aim is to take advantage of the flocculation of the aerated sludge itself or its supernatant (treated WW) to increase the dewaterability of the mixture through a flocculation improvement (Sanin & Vesilind 1994).

The last solution can be interesting punctually when the plant already exist with a septage source near the plant. Nevertheless, improving septage dewaterability with an activated sludge dilution has not been conclusive (Troesch et al., 2009b). This solution appears not economically interesting if we consider the surplus of equipment and work induced to mix them. Our latest researches (Troesch et al, 2009b; Vincent et al., 2010) are now related to direct treatment of septage by SDRB.

Septage characteristics

The septage characteristics exhibit high variations for most of the parameters (Table 4). This is due to different factors: varying ratio of sludge and liquid pumped by the vacuum truck, ratio tank volume/number of inhabitants and the emptying frequency. The high Capillarity Suction Time (CST) values and bound water percentage indicate a low dewaterability compared to activated sludge (typically at a level of approx. 7s). This poor dewaterability is partly due to the large particle size distribution of septage (d_{90}/d_{10}) and its high fines content (d_{10}) (see Fig. 4).

Table 4. Septage characteristics (in $mg \cdot L^{-1}$ except when indicated)

	DM	SS	VS (% SS)	COD	NK	NH4-N	PO4-P	CST (s)	d_{10} (mm)	d_{90}/d_{10}
Mean	33 700	27 000	68	47 000	1 555	302	46.0	398	3.3	21.1
Max.	70 500	64 000	79	87 000	2 460	441	59.6	841		

Sludge production

Estimating the sludge production of septic tank depends greatly of the emptying frequency of the tanks. On a 4 year storing into the septic tank we can estimate the septage production of about 6 kg of SS per year and per inhabitant with about 70 % of volatile solids (Liénard et al., 2008).

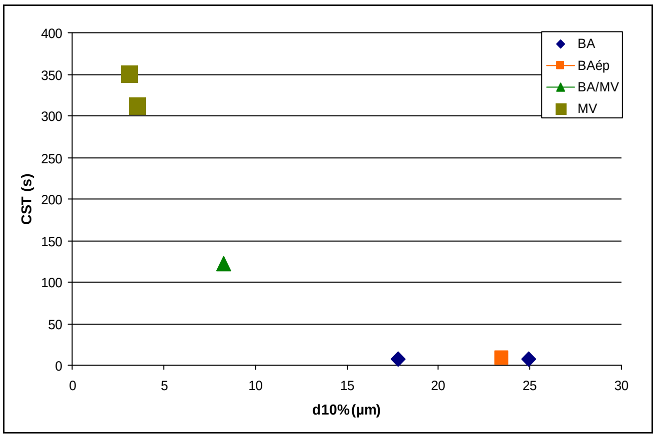

Figure 4. Influence of fine particles (express as d10) on CST.

Loads and performances

The loads tested were similar to the one applied with activated sludge.

- 30 kg of SS.m^{-2}.y^{-1} during commissioning period, using vegetal compost as filtration layer, allows a good reed development. Due to the low hydraulic load when working with septage, only 2 days of feeding is preferable during summer of the commissioning period to avoid water stress.
- Feeding the units at 50 kg of SS.m^{-2}.y^{-1} during normal functioning allows to maintain high DM content of the sludge at the end of the rest period (see Fig 5). DM content still always higher than 25 % and reach 70 % in summer.

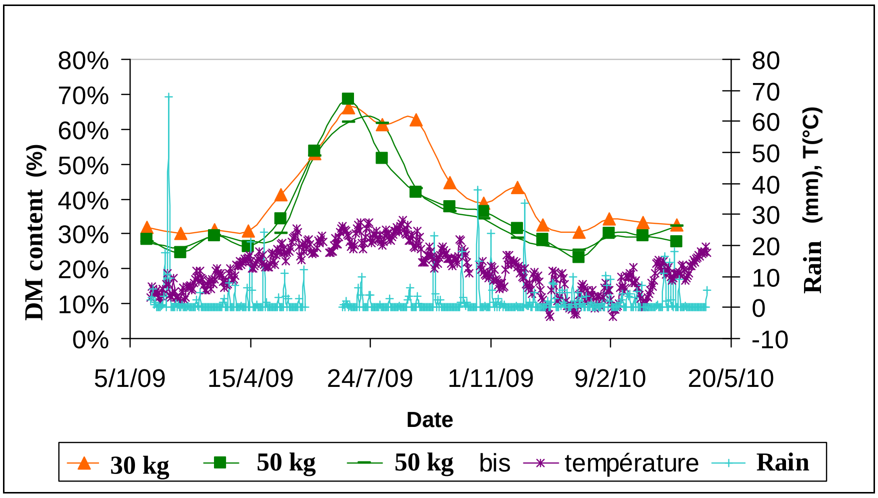

Figure 5. Evolution over the year of DM content of the sludge 25 days after the last feeding. Load applied 50 kg of SS.m^{-2}.y^{-1}.

When measuring oxygen content within the media and redox potential within the sludge, it appears that such a load is not problematic for the system. Actually load of 70 kg of SS.m^{-2}.y^{-1} are tested. On the other hand, the constraint when treating septage, is the lixiviate quality which needs a specific treatment. Indeed, despite of SS removal higher than 92 %, due to the high inlet concentration, the outlet SS concentration can reach values of 2 to 5 g·L^{-1} especially in commissioning period and in winter when short circuiting can appear due to sludge shrinkage cracks.

ACTUALS RESEARCHES

The actual researches done in Cemagref attend to better take into account the climatic conditions on design and operation modes of SDRB for the activated sludge treatment as well as for septage. This climatic impact is of importance on the one hand on sludge mineralization and sludge drying. The objectives are to build model to predict these kinetics.

- Mineralization kinetics are evaluated by using a original development of solid respirometric tool (Morvannou et al., 2010) to measured real sludge degradation into the units according to temperature.
- Sludge drying modelling necessitate to define the water stress coefficient of reeds.

BIBLIOGRAPHY

Lesavre J., Iwema, A. (2002). Dewatering of sludge coming from domestic wastewater treatment plant by planted sludge beds. 8th International Conference on Wetland systems for Water Pollution Control,Arusha, Tanzania.

Liénard A., Esser D., Deguin A. et Virloget F. (1990). Sludge dewatering and drying in reed beds : an interesting solution ? General investigation and first trials in France. Constructed Wetlands in Water Pollution ControlPergamon Press, Vol., 257-267.

Liénard A., Duchene P. et Gorini D. (1995). A study of activated sludge dewatering in experimental reed-planted or unplanted sludge drying beds. Water Science and Technology, 32(3), 251-261.

Liénard A. (1999). Deshydratation des boues par lits de sechage plantes de roseaux. Ingénierie EAT, 17, 33-45.

Liénard A., Canler J.P., Mesnier M., Troesch S. et Boutin C. (2008). Le traitement des matières de vidange: en station d'épuration ou en lits plantés de roseaux? Ingénieries EAT, 53, 35-48.

Morvannou A., Choubert J.M., Forquet N., Pilippe N., Vanclooster M., Molle P. (2010) Solid respirometry to characterise the biodegradation kinetics in VFCW : a better insight for modeling. In 12th International Conference on Wetlands systems for water pollution control, Venice, Italy.

Nielsen S. (2003). Sludge drying reed beds. Water Science and Technology, 48(5), 101-109.

Nielsen S. (2005). Sludge reed bed facilities: operation and problems. Water Science and Technology, 51(9), 99-107.

Sanin, F. D. & Vesilind, P. A. (1994). Effect of centrifugation on the removal of extracellular polymers and physical properties of activated sludge. Water Sci. Technol. 30(8), 117–127.

Torrens A., Molle P., Boutin C., Salgot M., (2006). Association of Stabilization Ponds and Intermittent Sand Filters: an appropriate wastewater treatment system for small communities, Proceedings, 7th International Conference on Small Water and Wastewater Systems, Mexico.

Troesch S., Liénard A., Molle P., Merlin G., Esser D. (2009a) Sludge drying reed beds: full- and pilot-scale study for activated sludge treatment. Water Science and Technology, 60(5), 1145-1154.

Troesch S., Liénard A., Molle P., Merlin G., Esser D. (2009b). Treatment of septage in sludge drying reed beds; a case study on pilot-scale beds. Water Science and Technology, 60(3), 643-653.

Vincent J., Molle P., Liénard A., Wisniewski C. (2010). Sludge drying reed beds for septage treatment: towards design and operation recommendations. In 12th International Conference on Wetlands systems for water pollution control, Venice, Italy.

Sludge drying reed beds in Italy

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Abstract
Several experiences about sludge treatment wetlands (TW) have been developed in Italy. In this paper data related to two different TWs in central Italy are reported. Biochemical, chemical-structural and toxicological aspects about sludge stabilization have been followed, in order to assess the efficiency of the process and to evaluate the toxicity of the final product in prevision of land application or its reuse in the environment. The efficiency of sludge treatment wetland in stabilizing the sludge organic matter was highlighted by the decrease in enzymatic activities and soluble forms of carbon and nitrogen. Moreover, the characterization of stabilised sludge by Pyrolysis-Gas Chromatography technique showed the formation of humic-like compounds over the time. The lack of toxicity of final product was demonstrated by the increase of germination index and of heavy metals bound to fractions linked with organic matter and mineral structures.

Keywords
Sewage sludge; wetland; organic matter; biochemistry; heavy metals; Pyrolysis-Gas Chromatography

INTRODUCTION
Sludge treatment wetlands (TW) have been successfully applied since the late 1980s in Europe (Uggetti et al., 2010). They are an alternative technology for municipal (Lienard et al., 1995; Nielsen, 2003) and agricultural sludges and slurries (Edwards et al., 2001). In Italy, one of the first pilot plants was developed in 1997 in a conventional wastewater treatment plant, where a load of 10 kg dw/m^2·y^{-1} was applied. The effectiveness of this pilot plant was monitored with particular attention to sludge dewatering and percolation water quality. Results were considered satisfying and sludge TW was recommended as an alternative to technological system of sludge dewatering (Barbieri et al., 2003). Similar projects were also developed for the dewatering of vinery sludges in northern Italy (Chiapella et al., 2003; Masi et al., 2005). These successful experiences have been used for the preparation of the guidelines about water treatment by the Institute for Environmental Protection and Research (Pucci et al., 2005) which indicates the sludge treatment wetland technology as a valid alternative for the dewatering of sludges.

CNR experience
A full scale implementation of TW has been carried out in Tuscany region (central Italy), since 2004 in several wastewater treatment plants managed by Acque S.p.A., one of the local holding companies for water and wastewater (Table 1). During the first period of investigation on a pilot plant, some parameters, correlated with the effectiveness of sludge treatment process, had been detected. These parameters allow following the processes of mineralization and humification of sludge organic matter in the reed beds from a chemical, biochemical and biological point of view. Moreover, these parameters correlated to the metabolic activity, organic matter evolution and biotoxicity, not only describe the biochemical processes occurring within the reed beds, but also allow to screen the stabilised sludge suitability in prevision for land application.

Cost analysis

The cost for transferred and treated offsite sewage sludges was estimated in about 12 €/m^3. Sludge TWs have been proved idoneous to treat on-site with a specific load of 3.8 m^3/m^2 of sludge per year and with an overall operation and maintenance cost estimated in about 4 €/($m^2 \cdot y$), or less for larger installations. Therefore, it is calculated a net saving for overall operation and maintenance costs of only 42€/($m^2 \cdot y$). Construction costs for reed beds are estimated in approximately 150 €/m^2, including the cost of emptying, transport and disposal of the treated sludge (the average cost of emptying, transport, and disposal of the treated sludge as agricultural amendment is estimated at 50 €/t in central Italy). These costs are supposed to be paid every 10 years, after the filling up of the reed beds. The yearly amortization amount of the construction costs, calculated at a nominal rate of 7%, reaches 21.4€/($m^2 \cdot y$). Thus, this technology provides a yearly net saving of 20.2 €/($m^2 \cdot y$).

MATERIALS AND METHODS

Sludge treatment wetland

Dewatering and stabilization of sludges were investigated in two urban wastewater treatment plants of Acque S.p.A. (Tuscany, Italy), WWTP 1 La Fontina, and WWTP 2 Oratoio, both near Pisa, (Tuscany Region, Italy) for 48 months. The drying beds present in both WWTPs were adapted and transformed into sludge treatment wetlands (TW).

The sludge treatment wetlands were composed by (1) eleven beds of 120 m^2 and (2) five beds of 75m^2 respectively, which were replicates. The beds were open (uncovered); the slope was 1% and each bed was provided with a drainage system made up of 2 layers of gravel with two different diameters: the bottom layer was 25 cm deep with gravel of 40/70 mm and the top layer was 15 cm deep with 5 mm gravel. Seedlings of *Phragmites australis* were planted at a distance of 50 cm x 50 cm during August 2005 and were irrigated with water effluent coming from the wastewater plant to enhance plant rooting. The outflow from the drainage system was collected by gravity and then pumped back to the treatment plant for further treatment.

Sewage sludge from conventional activated sludge was applied from October 2005 every 2 weeks from autumn to spring and every week during summer, corresponding to an annual rate of sludge of 38 kg dw/m^2 and 45 kg dw/m^2, respectively.

Table 1. Wastewater treatment plants.

WWTP	Basin area (m^2)	Loading rate (kg dw/$m^2 \cdot y$)	Sludge (% dw)	Treated sludge (m^3/y)
La Rotta (400 p.e.)	100 (4 beds)	60	1%	400
La Fontina (30000 p.e)	1210 (11 beds)	38	1%	4600
Oratoio (10000 p.e.)	375 (5 beds)	45	1%	1700
Pittini (5000 p.e.)	256 (6 beds)	67	1.5%	1200
Colle di Compito (4000 p.e.)	225 (5 beds)	67	1.5%	1000
Stabbia (3000 p.e.)	64 (3 beds)	67	1.5%	300

Methods

The following parameters related to different aspects are reported:

Sludge Mineralization: Total organic carbon (TOC), total nitrogen (TN), ammonia, water-soluble carbon (WSC), dehydrogenase (Dhase), β-glucosidase and urease activities.

Sludge Humification: Fulvic acids (FA), humic acids (HA), index of mineralization (O/N) and humification (B/E3) derived by pyrolysis–gas chromatography (Py-GC) analyses.

Sludge Biotoxicity: Germination index (GI), heavy metal fractionation.

TOC and TN were analysed by RC-412 multiphase carbon and FP-528 protein/nitrogen (Leco). WSC, fulvic and humic acids were determined according to the method of Yeomans and Bremner (1988). Ammonium was determined with the ammonia-selective electrode SevenMulti (Mettler Toledo). Dhase activity was tested by the method of Masciandaro (2000), using 2-p-iodo-nitrophenylphenil-tetrazolium (INT) as substrate. β-glucosidase and urease activities were measured according to Garcia et al. (1993) methods. The dried sludges were put into pyrolysis microtubes in a platinum coil probe (CDS Pyroprobe 190) and pyrolysis was carried out at 800°C for 10 s, with a heat gradient of 108C/ms. The probe was directly coupled to a Carlo Erba 6000 gas chromatograph with a flame ionization detector (FID). The pyrograms obtained were quantified by normalizing the areas of the selected seven peaks (acetic acid, acetonitrile, benzene, toluene, furfural, pyrrole and phenol). Identification of pyrogram fragments in the samples was carried out comparing the relative retention times to standard spectra (Ceccanti et. al, 2007). The germination index was determined on aqueous extract following the Hoekstra et al., method (2002). For heavy metal fractionation the Community Bureau of Reference (BCR) method was followed (Mocko and Waclawek 2004).

RESULTS

Sludge mineralization

Organic matter content (TOC and TN) decreased in TW 1, while it increased in TW 2. The mineralization of organic matter was highlighted in both TWs by the impressive decrease of dehydrogenase activity, which expresses the biological oxidation processes of microbial activity. The same can be also said for the significant decrease of soluble forms of carbon (WSC) and nitrogen ($N\text{-}NH_4^+$) and their correlated enzymatic activities (β-glucosidase and urease) (Fig. 1). All of the biochemical activities and their soluble forms, in fact, reached values that kept metabolic activity of the sludge at low levels in both wastewater plants, even if sewage sludges were continuously added to the reed beds. This could suggest that the organic matter is approaching a bio-stabilization status.

Table 2. Total Organic Carbon (TOC, %), Total Nitrogen (TN, %) and Dehydrogenase activity (Dhase, mg INTF/kg dw h) in TW 1 and TW 2.

	TOC (%)		TN (%)		Dhase (mg INTF/kg dw·h)	
months	TW 1	TW 2	TW 1	TW 2	TW 1	TW 2
6	30.2 a	31.2 c	5.20 a	4.84 a	213 a	174 a
12	28.1 b	25.7 d	3.77 bc	3.41 b	13.6 c	21.9 b
24	29.3 a	24.3 d	4.79 ab	3.32 b	31.5 b	9.45 c
36	28.0 b	34.0 b	3.91 b	4.86 a	3.41 c	27.1 b
48	28.3 b	38.8 a	3.55 c	4.16 ab	7.65 c	16.2 c

Sludge humification

Humic substances are commonly constituted by two different components: fulvic acids, which represent the less stable part of humic matter and humic acids, which represent the more stable fraction of humic matter. Fulvic acids decreased significantly in both TWs during the time, while humic acids remained quite unchanged, with higher values after 48 months (Fig. 2). In order to complete the picture of the mineralization versus humification trends of the sludge organic matter Pyrolysis-Gas Chromatography (Py-GC) technique has been used.

Py-GC enables the characterization of soil organic matter quality from a chemical-structural point of view. The index of humification B/E3 (benzene to toluene, the former deriving basically from condensed aromatic structures, the latter deriving from aromatic structures containing short aliphatic chains) increases when organic matter is becoming more mature. The index of mineralization O/N expresses the ratio between pyrrole (a heterocyclic aromatic organic compound derived from nitrogenous compounds, humified organic matter, and microbial cells), and furfural (a pyrolytic product coming from polysaccharides degradation). The higher the ratio, the higher the extent of mineralization of soil organic matter, meaning that just a low concentration of labile organic compounds, like polysaccharides, is still present. The mineralization index (O/N) showed in both TWs a tendency to decrease, while the humification index constantly increased, in particular in TW 1 (Fig. 2); these results confirmed the effectiveness of this biostabilization process, in that as the labile organic matter is degraded the more stable one (with humic characteristics) is being formed during the time (Ceccanti et al., 2007).

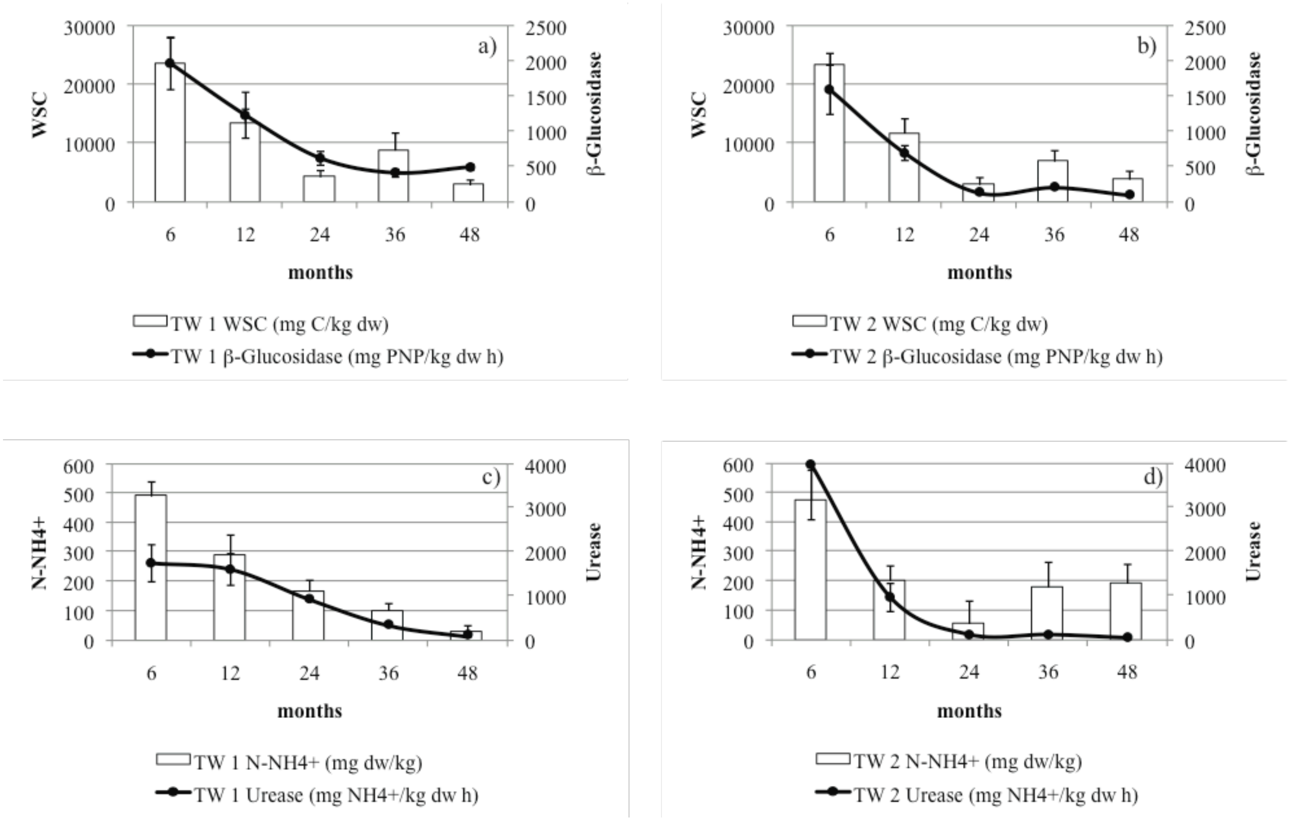

Figure 1. Water Soluble Carbon (WSC, mg C/kg dw) and β-Glucosidase activity (b-Glucosidase, mg PNP/kg dw h) in TW 1 (a) and in TW 2 (b). Ammonia ($N\text{-}NH_4^+$, mg/kg dw) and Urease activity (mg NH_4^+/kg dw h) in TW 1 (c) and in TW 2 (d).

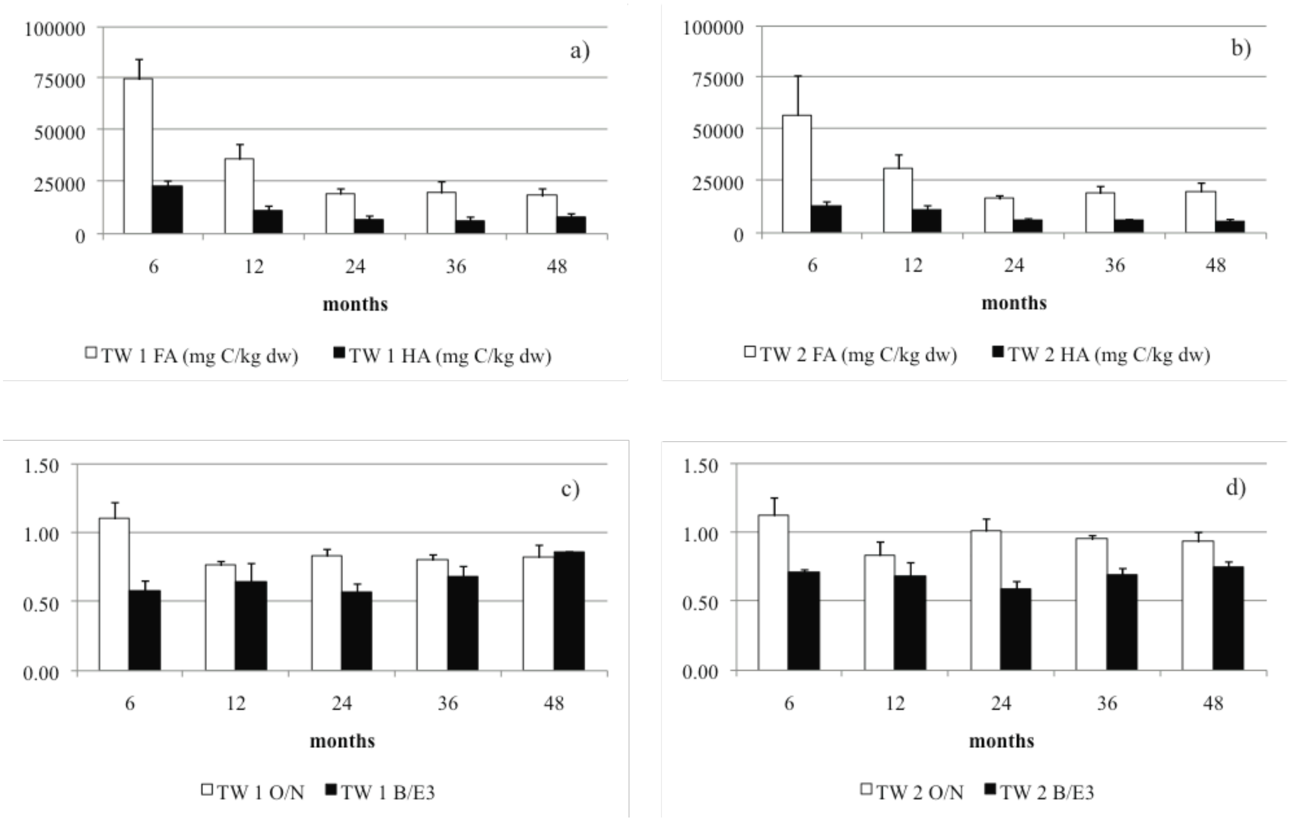

Figure 2. Fulvic acids (FA, mg C/Kg dw) and humic acids (HA, mg C/kg dw) in TW 1 (a) and in TW 2 (b). Pyrolytic indices of mineralization (O/N) and humification (B/E3) in TW 1 (c) and in TW 2 (d).

Sludge toxicity

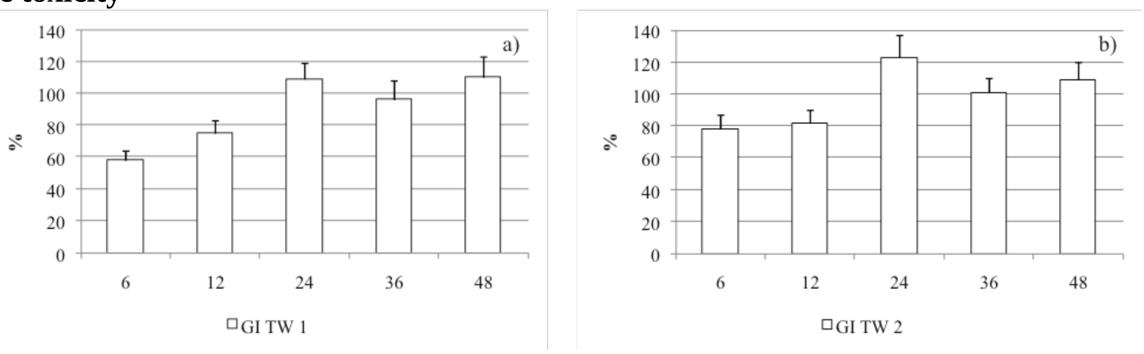

Figure 3. Germination index (GI, %) in TW 1 (a) and in TW 2 (b).

The germination index (GI) evaluates the toxicity of a material (compost, soil or other organic matrices) in prevision of agricultural or environmental uses. Values below 40% indicate high toxicity, lethal to vegetal species; values between 40 and 60% indicate toxicity still capable of causing damage, but not lethal; values above 60% do not cause damage to plants. In our experiments, GI increased significantly over time in both TWs (Fig. 3) and the values obtained did not indicate problems of plant toxicity (Fuentes et al. 2006).
The determination of the content of heavy metals and their fractionation give information about a particular type of toxicity which is more worrying for ecological aspects and human health.
The total heavy metal content (Table 3) remained below the level established by law for the reuse of sewage sludge in agriculture (Italian regulation ex D. Lgs. 99/92). Moreover, the metal content remained stable during the time with some fluctuations, thus suggesting that a relative increase of heavy metal concentration was not observed, probably due to their phytoextraction by plants.

Table 3. Total heavy metal content in TW 1 and TW 2.

	Cr (mg/kg dw)		Cu (mg/kg dw)		Ni (mg/kg dw)		Cd (mg/kg dw)		Pb (mg/kg dw)		Zn (mg/kg dw)	
months	TW 1	TW 2	TW 1	TW 2	TW 1	TW 2	TW 1	TW 2	TW 1	TW 2	TW 1	TW 2
6	29a	40a	362a	467a	67a	55a	2.7	< 2	91a	121a	917a	1357a
12	59b	73b	410b	467a	34b	28b	< 2	< 2	93a	100a	1297a	1535a
24	123c	384c	573c	360a	52a	26b	2.53	< 2	86a	39b	1836b	915b
36	24a	102b	573c	608b	32b	26b	< 2	< 2	99a	70ab	1538b	1462a
48	55b	159b	549c	563b	34b	30b	< 2	< 2	43b	30a	1705b	1532a
D. Lgs. 99/92	-		1000		300		20		750		2500	

The procedure for fractionation differentiated the sludge heavy metals into four fractions:

1. Exchangeable fraction associated with carbonated phase (Fraction 1). Metals are adsorbed on the sludge components and Fe and Mn hydroxides. This is the most mobile fraction potentially toxic for plants.
2. Reducible fraction associated with Fe and Mn oxides (Fraction 2). Heavy metals are strongly bound to these oxides but they are thermodynamically unstable in anoxic and acidic conditions.
3. Oxidisable fraction bound to organic matter (Fraction 3). It is well known that metals may be complexed by natural organic substances. These forms become soluble when organic matter is degraded in oxidising conditions. This fraction is not considered to be bioavailable and mobile because the metals are incorporated into stable high molecular weight humic substances, which release small amounts of metals very slowly.
4. Residual fraction (Residual Fraction). The residual solids mainly contain primary and secondary solids that occlude the metals in their crystalline structures. They are considered to be not extractable and in an inert form.

In order to understand results about fractionation of stabilised sludge from TWs, a sludge from storage pond, where sludge was stored for three years without any additional treatment, was used as control (Sludge Control). Results about fractionation showed that the sludge phytostabilization, activating the humification process, enabled heavy metals to link tightly with humified organic matter, thus making them less available in the environment after 48 months of treatment (Fig. 4). In fact, most of all metals were linked to fractions less available to vegetation, such as Fraction 3 (bound to organic matter) and Residual Fraction (related to the mineral structure). Other authors have found higher content of metals associated with Fraction 1 and Fraction 2 in sewage sludge stabilised with traditional methods (thickening, heat treatments, anaerobic digestion, etc) (Walter et al. 2006; Fuentes et al., 2008). Moreover, by the comparison with the untreated control sludge, it appears that sludge phytotreatment greatly reduced Fractions 1 and 2 in both TWs ($p < 0.05$).

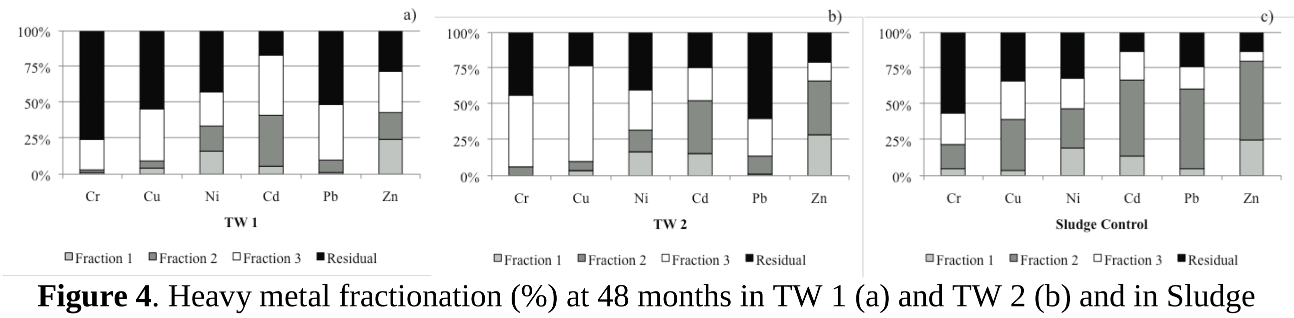

Figure 4. Heavy metal fractionation (%) at 48 months in TW 1 (a) and TW 2 (b) and in Sludge control (c).

Statistical analysis

Principal component analysis (PCA) was applied to all results obtained during the experiments. The PCA is a multivariate statistical data analysis technique which reduces a set of raw data to a number of principal components that retain the most variance within the original data in order to identify possible patterns or clusters between objects and variables (Fuentes et al., 2008; Pardo et al., 2004).

In order to minimize the number of variables, the contents of different metal fraction were transformed from mg/kg to meq/kg and then the content for each fraction was summed. The PCA of the data set indicated 69.9% of the data variance as being contained in the first two components (Fig. 5, a). PC 1 and PC 2 account for 52.0%, and 17% of the total variance. PC 1 was closely associated with mineralization (WSC, $N\text{-}NH_4^+$, enzymatic activities, fulvic and humic acids) and toxicity parameters (Fraction 1, Residual fraction and germination index), while PC 2 was linked with humification and organic matter parameters (TOC, humification index B/E3 and Fraction 3).

Both TWs had a similar behaviour over the time (Fig. 5, b): they were situated at the bottom right of the plot at the beginning, then they shifted towards PC1 and PC2 to the top left of the plot. At first, the two TWs were linked with mineralization (soluble nutrients, enzymatic activities) and toxicity parameters (Fraction 1 and Fraction 2), while at the end of experimentation they resulted associated with humification index, germination index, Fraction 3 and Residual fraction.

a)

	PC 1	PC 2
TOC	0.08	0.87*
TN	0.51	0.62
WSC	0.95*	0.05
$N\text{-}NH_4^+$	0.87*	0.19
Dhase	0.81*	0.25
Urease	0.89*	-0.25
β-glucosidase	0.83*	-0.15
Fulvic acids	0.98*	0.10
Humic acids	0.92*	-0.13
O/N (Py-GC)	0.48	0.44
B/E3 (Py-GC)	-0.28	0.64*
Germination index	-0.88*	0.20
Fraction 1	0.86*	-0.05
Fraction 2	0.32	-0.41
Fraction 3	0.05	0.69*
Residual Fraction	-0.71*	-0.12
Explained variance	8.29	2.64
Total proportionality	0.52	0.17

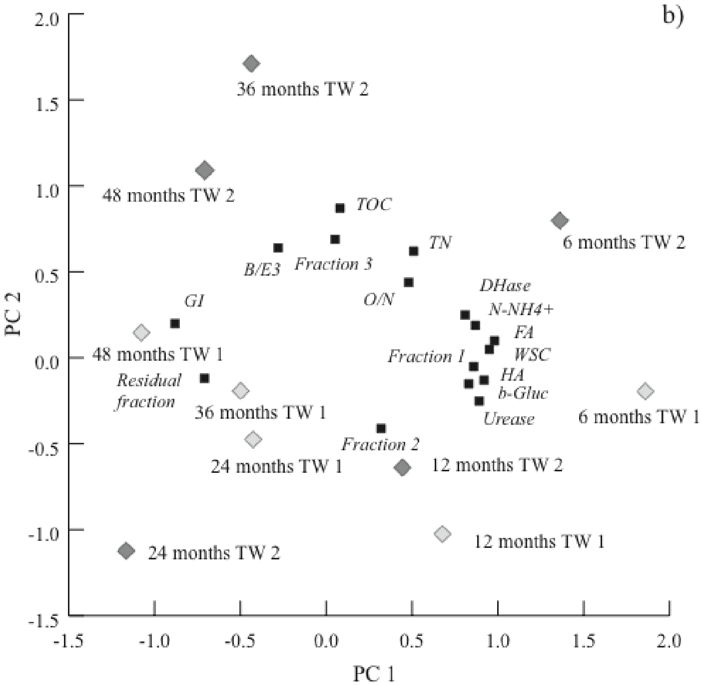

Figure 5. Principal component analysis: a) Principal component loadings. *parameters used for PCA interpretation; b) Biplot of scores and loadings (PC 1 vs PC 2).

CONCLUSIONS

The results of biochemical analyses demonstrated the efficiency of the sludge treatment wetland in stabilizing the sludge organic matter, as shown by the decrease in enzymatic activities and soluble forms of carbon and nitrogen. Moreover, the characterization of stabilised sludge from the chemical-structural point of view highlighted the formation of humic-like compounds over the time. In addition, the results of germination index and heavy metal fractionation demonstrated that the final product is not toxic and, for this reason, it could be used in agriculture or environmental practices. In fact, the stabilization process in reed bed systems helped the immobilization of heavy metals, in that by comparison with the untreated control sludge, it appears that heavy metals bound to organic matter (Fraction 3) and mineral structures (Fraction 4) were greatly increased in both TWs.

REFERENCES

Barbieri A., Garuti G., Avolio F., Bruni S., (2003). Sludge dewatering using macrophytes in a small wastewater system: a case study of a pilot scale plant in northern Italy. *Journal of Environmental Science Health, A*, **38**, 2425-2433.

Ceccanti B., Masciandaro G., Macci C. (2007). Pyrolysis-gas chromatography to evaluate the organic matter quality in a mulched soil. *Soil and Tillage Research*, **97**, 71-78.

Chiapella P., Cortemiglia C., Garuti G. (2003). Esperienze di fitodisidratazione dei fanghi di depurazione di un'azienda vinicola, *L'Ambiente*, **5**, 30-34.

Edwards J.K., Gray K.R., Cooper D.J. Biddlestone A.J., Willoughby N., (2001). Reed bed dewatering of agricultural sludges and slurries. *Water Science and Technology*, **44**, 551-558.

Fuentes A., Llorens M., Saez J., Aguilar M. I., Ortuno J. F., Meseguer V. F. (2008). Comparative study of six different sludges by sequential speciation of heavy metals. *Bioresource Technology*, **99**, 517-525.

Garcia C., Hernandez T., Costa F., Ceccanti B., Masciandaro G. (1993). A study of biochemical parameters of composted and fresh municipal wastes. *Bioresource Technology,* **44**, 17-23.

Hoekstra N.J., Bosker T., Lantiga E.A. (2002). Effects f cattle dung from farms with different feeding strategies on germination and initial root growth of cress (*Lepidium sativum* L.), *Agriculture, Ecosystems and Environment,t* **93**, 189-196.

Lienard A., Duchene P., Gordi D. (1995). A study of activated sludge dewatering in experimental reed-planted or unplanted drying beds. *Water Science and Technology,* **32**, 251-261.

Masciandaro G., Ceccanti B., Ronchi V., Bauer C. (2000). Kinetic parameter of dehydrogenase in the assessment of the response of soil to vermicompost and inorganic fertilisers. *Biology and Fertility of Soils*, **32**, 479-483.

Masi F., Conte G., Martinuzzi N., Bresciani R. (2005). Progetto Cantina sociale di Casorzo e zone limitrofe. http://www.iridra.com (access April 2010)

Mocko A., Waclawek W. (2004). Three-step extraction procedure for determination of heavy metals availability to vegetables. *Analytical and Bioanalytical Chemistry,* **380**, 813-817.

Nielsen S. (2003). Sludge drying reed beds. *Water Science and Technology,* **49**, 101-109.

Pardo R., Helena B.A., Cazurro C., Guerra C., Deban L., Guerra C.M., Vega M. (2004). Application of two-and three-way principal component analysis to the interpretation of chemical fractionation results obtained by the use of the B.C.R. procedure. *Analytica Chimica Acta*, **523**, 125-132.

Pucci B., Masi F., Conte G., Martinuzzi N., Bresciani R., " Linee Guida per la Progettazione e Gestione di Zone Umide Artificiali per la Depurazione dei Reflui Civili", a cura di M. Mazzoni, APAT – ARPAT, Firenze. 2005

Uggetti E., Ferrer I., Llorens E., Garcia J. (2010). Sludge treatment wetlands: A review on the state of the art. *Bioresource Technology,* **101**, 2905-2912.

Walter I., Martinez F., Cala V. (2006). Heavy metal speciation and phytotoxic effects of three representive sewage sludges for agricultural uses. *Environmental Pollution*, **139**, 507-514.

Yeomans J.C., Bremner J.M. (1988). A rapid and precise method for routine determination of organic carbon in soil. *Communication in Soil Science and Plant Analysis,* **19**, 1467-1476.

Economic analysis and life cycle approach to compare drying reed beds and conventional treatments for sludge management

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Abstract
Sludge treatment wetlands (STW) emerge as a promising sustainable technology with low energy requirements and operational costs. In this work economic and environmental aspects of STW are investigated to compare alternatives for sludge management. To this end, cost analysis and life cycle assessment (LCA) were carried out considering dimensions and operation criteria of full-scale systems located in Spain. Four scenarios are considered: 1) STW with direct land application, 2) STW with compost post-treatment, 3) centrifuge with compost post-treatment and 4) sludge transport to an intensive wastewater treatment plant. The economic analysis shows that in small facilities (500-2000 PE), constructed wetlands with direct land application is the most favorable solution (less than 0.15 €/m^3 of water treated). The costs are slightly increased if post-treatment is required (between 0.18 and 0.15 €/m^3). On the other hand, centrifugation costs decrease at increasing wastewater flow rates, as a result of high implementation costs (from 0.28 to 0.15 €/m^3). According to the SimaPro LCA, STW with direct land application correspond to the solution with lower environmental impact. In all scenarios global warming is a significant impact category, which is attributed to fossil fuel and electricity consumption; while greenhouse gas emissions from STW are insignificant. On the whole, STW is the most appropriate solution to manage waste sludge produced in decentralized and small communities (<2000 PE), mainly post-treatment is not required prior to land application.

Keywords
Constructed wetlands; biosolids; composting; drying reed beds; sanitation, wastewater

INTRODUCTION

A major concern of intensive sewage treatment processes is the large production of waste sludge, which is generally managed by complex and costly operations. Its production is highly variable depending on the wastewater treatment used, for instance conventional activated sludge processes produce from 60 to 80 g of total solids (TS) per person per day (Von Sperling and Gonçalves, 2007). During the last years, sludge generation has increased dramatically by the fast growth of world population and industrialisation (Hong et al., 2009). According to Fytili and Zabanitou (2008), sludge production has increased in the European Union by 50 % since 2005. Therefore, optimisation of sludge management becomes a key element in the wastewater treatment sector.

Conventional sludge stabilisation and dewatering technologies (i.e. anaerobic digestion followed by centrifugation or filtration) are costly and energy demanding, which is troublesome particularly in small facilities (<2,000 population equivalent (PE)). This is a matter of concern, since the number of small wastewater treatment plants (WWTP) in operation will continue to increase within the next years, including municipalities below 500 PE (Council of the European Union, 2000). Nowadays, the solution adopted in many small facilities is sludge transport to the nearest WWTP with a conventional sludge treatment line, posing high operation costs and high potential environmental impacts. In this context, simplified *in situ* treatments are needed.

Sludge treatment wetlands (STW) consist of shallow tanks filled with a gravel layer and planted with emergent rooted wetland plants such as *Phragmites australis* (common reed). Sludge is spread and stored on the surface of the beds where most of its water content is lost by evapotranspiration of the plants and by water draining through the gravel filter layer, leaving a concentrated sludge residue on the surface. When the maximum storage capacity is reached, after a final resting period, the final biosolids are withdrawn to start a new operating cycle. Evolution of sludge composition results from dewatering and mineralisation processes (Nielsen, 2003). The resulting final product is suitable for land application (Nielsen and Willoughby, 2005); although in the practice in some cases it is post-treated to improve sludge stabilisation and hygienisation (Zwara and Obarska-Pempkowiak, 2000).

In comparison with common mechanical dewatering technologies like centrifuges, sludge treatment wetlands emerge as a promising alternative (Uggetti et al., 2010), which has low energy requirements, reduced operation and maintenance costs, and in principle causes little environmental impact. However, a systematic evaluation of the environmental performance of this technology has not yet been reported.

In this study, STW costs and environmental impact are investigated and compared to conventional treatments for sludge management in small communities (<2,000 PE). Economic and environmental assessments have been carried out assuming design and operation criteria of full-scale systems located in Spain. Four scenarios are considered and compared: 1) STW with direct land application of the final product, 2) STW with compost post-treatment, 3) centrifugation with compost post-treatment, 4) sludge transport to an intensive WWPT without previous treatment. Our aim was to demonstrate the suitability of STW for small communities, not only in terms of process performance but also in terms of costs and environmental impacts.

MATERIALS AND METHODS

Economic assessment

Economic aspects of STW are compared with sludge management alternatives which are currently used in small WWTP in our zone: centrifugation, as representative of mechanical dewatering techniques, and transport to a larger WWTP with sludge treatment line. According to the common practice adopted nowadays in Spain, STW followed by composting is also considered (scenario 2). Each scenario is evaluated for sewage treatment capacities of 100, 200 and 400 m^3/d of wastewater treated, theoretically corresponding to 500; 1,000 and 2,000 PE.

According to design and operation criteria of STW located in Spain (Uggetti et al., 2009), we considered between 4 and 12 beds, with an average surface of 50 m^2 and height of 1.6 m. Taking into account the 20 cm layer of gravel and sand, the sludge storage capacity results in 50 m^3. In this study, sludge loading rate of 50 kg TS/m^2·year and 5 year operating cycles are assumed, although longer operating cycles are reported in other countries like Denmark (Nielsen, 2003). Emptying procedures involve final biosolids withdrawal with a power shovel and transport to final destination. STW operation is thereafter re-started without replanting.

Table 1 summarises sludge flow rates for each scenario. Secondary sludge generation in the WWTP is calculated by the Huisken equation. The difference between sludge production in STW and centrifuge is due to the TS concentration of the final product, 25 and 20 % TS, respectively (Uggetti et al., 2010). CH_4 emissions were measured by gas chromatography (Thermo Finnigan Trace, GC 2000) in samples collected from representative STW by positioning a Linvall Hood of 1 m^2 surface area as described by Sarkar and Hobbs (2002).

Table 1. Sludge flow rates considered in the economic and environmental assessment.

	Wastewater treated		
	100 m^3/d	**200 m^3/d**	**400 m^3/d**
Waste activated sludge (sludge generation) (m^3/year) (all scenarios)	275	550	1100
Sludge production in STW (m^3/year) (scenarios 1 and 2)	33	66	132
Sludge production in centrifuge (m^3/year) (scenario 3)	41	82	165
Pump electricity consumption in STW (kWh/year) (scenarios 1 and 2)	25	50	105
Pump electricity consumption in centrifuge (kWh/year) (scenario 3)	30	60	125
Centrifuge electricity consumption (kWh/year) (scenario 3)	140	280	560
CH_4 emission rate from STW ($mg/m^2 \cdot s$) (scenarios 1 and 2)	< 88	< 88	< 88

Life cycle assessment

The aim of the LCA model developed is to compare the environmental impact of STW with sludge management alternatives commonly used in small WWTP in our zone. Therefore, the same scenarios as in the economic analysis are considered.

The function of the system is to manage secondary sludge produced in an activated sludge unit with extended aeration. For this reason, the functional unit is defined as the management of 1 ton of sewage sludge (wet weight).

Since the study is focused on sludge management, secondary sludge is selected as input material; and only the impact generated by sludge management in the facility is accounted for. This includes the sludge treatment line of the WWTP (STW or centrifuge) and transport to post-treatment in a composting plant (scenarios 2 and 3) or treatment in an intensive WWTP (scenario 4), assuming a distance of 30 km in all cases. Treatments outside the WWTP (composting in scenarios 2 and 3; and sludge treatment in a larger WWTP in scenario 4) are not included in the model.

Raw materials required for systems' construction and energy consumption for systems' operation are taken into account. On the contrary, the systems' boundaries exclude the construction phase, which only accounts for minor environmental impacts compared to the operation phase of WWTP, according to previous LCA studies (Lundie et al., 2004 and Lassaux et al., 2007). The end of life is included for the centrifuge, as it should be replaced over the period considered (20 years); but not for STW, since their lifespan is longer than the 20 years period considered in this study.

Inventory data on systems' design and operation are the same as for the economic analysis. Data concerning the embodied environmental aspects of materials, transport use and other processes were taken from the Ecoinvent system process database. The LCA analysis was carried out with the software SimaPro 7.1 by PRé Consultant, using the CML 2 baseline method (Guinée, 2001). Impact categories evaluated include Abiotic Resource Depletion, Acidification, Eutrophication and Global Warming Potential (Climate Change), amongst others.

RESULTS

Economic assessment

Table 2 shows investment and operation costs of scenarios 1, 2 and 3 (scenario 4 does not have investment costs, hence it is not included). The results are expressed in €/m^3 of wastewater treated. STW investment costs include soil occupation and excavation, wetlands construction, pump and pipe installation, gravel placement and plantation. The most significant costs of the centrifuge include machine assembly and installation, room construction and polyelectrolyte preparation. Notice that STW investment costs increase with the treatment capacity, from 60,000 to 190,000 € for 500 and 2,000 PE systems, respectively. On the other hand, centrifuge costs increase only slightly, from 90,000 to 120,000 €. Therefore, the difference between investment costs becomes more evident for 2,000 PE facilities and are more competitive for centrifuges.

Table 2. Investment and operation costs of scenarios 1, 2, 3 and 4: (1) STW, (2) STW + compost, (3) centrifuge + compost and (4) transport to WWTP. Costs are expressed in €/year.

		Wastewater treated		
		100 m^3/d	**200 m^3/d**	**400 m^3/d**
Scenario 1	Investment cost	60.169	99.490	189.736
	Operation cost	4.012	7.259	13.199
Scenario 2	Investment cost	60.169	99.490	189.736
	Operation cost	5.007	9.237	17.161
Scenario 3	Investment cost	88.722	90.448	114.950
	Operation cost	7.952	13.564	20.939
Scenario 4	Investment cost	-	-	-
	Operation cost	11.348	21.018	38.430

The economic analysis considering a life cycle of 20 years is shown in Fig. 1. It is calculated assuming 3% increase of operation costs and applying 5% interest tax to the total cost. In this case, amortisation of investment and STW emptying costs are also included. From a long term perspective, the benefit of biosolids' direct land application (scenario 1) emerges versus compost post-treatment (scenario 2), with lower costs (0.021 €/m^3) in all cases. Investment and operation costs of the centrifuge (0.28 €/m^3) are more expensive than other solutions (0.24 €/m^3 for transport and 0.16-0.18 €/m^3 for STW) for communities of 500 PE. However, centrifugation costs decrease at increasing treatment capacity (to 0.20 and 0.15 €/m^3 for 1,000 and 2,000 PE systems, respectively), hence treatment costs are the same as STW for 2,000 PE systems. Transport may be considered as an alternative to centrifugation only for systems with less than 850 PE (0.28 €/m^3 versus 0.24 €/m^3). Likewise, STW costs are 0.05-0.07 €/m^3 lower than this option. It is worth mentioning that the economic evaluation of this scenario is correlated with sludge production (and humidity), as well as the distance to nearest WWTP with sludge treatment line. In this study, an average distance of 30 km was adopted, based on circumstances normally observed in our zone.

This analysis underlines the economic advantage of STW with respect to conventional treatments exemplified by centrifugation in facilities up to 2,000 PE. However, this technology is currently adopted for sludge management in systems up to 30,000 PE in Italy (Peruzzi *et al.*, 2007) and 60,000-125,000 PE in Denmark (Nielsen, 2003). Certainly, the results are specific for each country, depending on the costs (i.e. electricity), as well as design and operation criteria of STW and weather conditions, affecting the efficiency of the treatment. For instance, operating cycles of 5 and 10 years are described in Spain and Denmark, respectively. Longer operating cycles reduce operation costs of STW, resulting in additional economic advantage for communities above 2,000 PE.

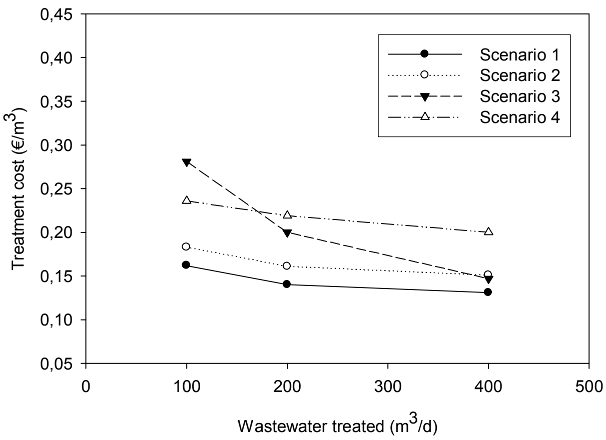

Figure 1. Investment and operation costs over 20 years of operation of scenarios 1, 2, 3 and 4: (1) STW, (2) STW + compost, (3) centrifuge + compost and (4) transport to WWTP.

Life cycle assessment

In LCA analysis the environmental impacts attributed to materials or processes are grouped according to the so-called impact categories. LCA results are therefore expressed as a quantification of the potential contribution of materials and processes to each impact category. Fig. 2 shows the main impact categories of this LCA model (Abiotic Resource Depletion, Acidification, Eutrophication and Global Warming Potential (Climate Change)), with comparative results for each scenario. The results are presented in absolute values in the units corresponding to each impact category. Within each impact category, the total impact as well as the individual contribution of raw materials, energy and transport are included separately. This interpretation is useful to determine the most influent element of the process that could eventually be modified to reduce the global impact.

In general, within each category the total impact is distributed following the same pattern: transport (scenario 4) has the highest impact, from 3 to 6 times higher than centrifuge with compost post-treatment (scenario 3) and STW with compost post-treatment (scenario 2). The impact of STW with direct use of the final product (scenario 1) is negligible in comparison with the other scenarios, with values between 1,000 and 6,000 times lower. According to this analysis, STW appear as the most favourable solution in every impact category. For this scenario 1 the biggest impact is caused by raw materials employed in system's construction; while direct greenhouse gas emissions (Table 2), as well as indirect emissions derived from energy consumption and transport, have a smaller contribution. On the whole, STW impact is negligible in comparison with the rest. If post-treatment is required, the total impact of STW (scenario 2) and centrifuge (scenario 3) is similar, due to sludge transport to post-treatment. From an environmental point of view, centrifuges and filter bands do not have relevant differences (Gallego *et al.*, 2008), therefore scenario 3 should be representative of conventional mechanical dewatering treatments.

Global Warming Potential accounts for a high contribution mainly in scenarios 2, 3 and 4 (1,100; 1,300 and 6,000 kg CO_2eq/t wet weight, respectively) due to fossil fuel and electricity consumption. In STW, the contribution of CH_4 emissions to this impact category is negligible, as a result of the low CH_4 found in these type of systems (Table 1).

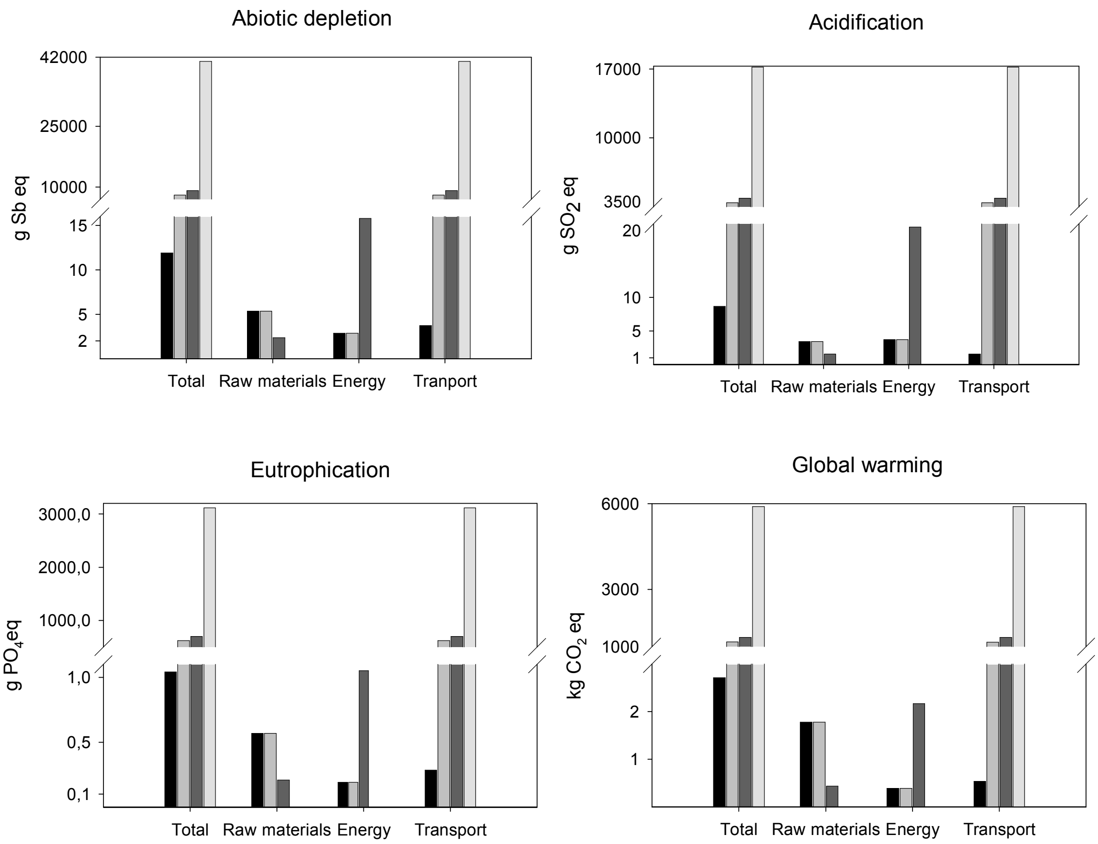

Figure 2. Life Cycle Assessment results grouped by impact categories for scenarios 1, 2, 3 and 4: (1) STW, (2) STW + compost, (3) centrifuge + compost and (4) transport to WWTP.

If we look at individual contributions of raw materials, energy and transport within each scenario (Fig. 2), other trends are observed. Scenario 1 is characterised by a high consumption of raw materials (basically steel and gravel), which accounts for the highest contribution in all impact categories. On the other hand, lower impacts are attributed to the energy consumption for sludge pumping into the STW, and transport during STW emptying operation.

Scenario 2 has the same contribution as scenario 1 with respect to raw materials and energy, but in this case transport accounts for the highest impact, which is attributed to the compost post-treatment. In scenario 3, the centrifuge has low raw materials requirements, but significantly higher energy consumption for sludge dewatering and pumping. Like in scenario 3, transport to compost post-treatment has the highest contribution to the total impact. As in the economic study, sludge transport to an intensive WWTP (scenario 4) is characterised by the highest environmental impact in all categories. Indeed, the reduction of sludge volume after dewatering (scenarios 1-3) has a positive environmental impact with respect to untreated sludge transport.

The results of this assessment show the economic and environmental benefits of STW compared to conventional mechanical dewatering and transport of untreated sludge. STW are less advantageous if compost post-treatment is required, as with mechanical dewatering techniques, due to the impact associated to sludge transport. However, the impacts of composting may differ between partially stabilised sludge from STW and dewatered sludge from centrifuges. For this reason, further LCA studies should include the post-treatment stage as well as final disposal of biosolids. As indicated by Cambell (2000), the most important criterion in the selection of sludge management alternatives is that the solution must be appropriate to the local conditions of each site.

CONCLUSIONS

This study looked at economic and environmental aspects of sludge treatment wetlands for small communities (500-2,000 PE). From this evaluation, STW with direct land application emerge as the most cost-effective scenario, which is also characterised by the lowest environmental impact (almost negligible in comparison with the other options evaluated). The LCA highlights that in all scenarios global warming has a significant impact, which is attributed to fossil fuel and electricity consumption; while methane emissions from STW are insignificant. As a conclusion, sludge treatment in constructed wetlands with direct land application is the most appropriate solution to manage waste sludge in decentralised small communities.

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REFERENCES

Council of the European Union (1986). Council Directive 86/278/EEC of 12 June 1986 on the Protection of the Environment, and in Particular of the Soil, when Sewage Sludge is Used in Agriculture. *Official Journal of the European Union* L 181, 04/07/1986, 6-12.

Fytili, D. and Zabaniotou A. (2008). Utilization of sludge in EU application of old and new methods – A review. *Renewable & Sustainable Energy Reviews* 12, 116-140.

Gallego, A., Hospido, A., Moreira, M.T., Feijoo, G. (2008). Environmental performance of wastewater treatment plants for small populations. *Resources conservation and recycling* **52**, 931-940.

Guinée, J. (2001). Life Cycle Assessment: an operational guide to ISO standards. Centre of Environmental Science, Leiden University, Kluwer Academic Publisher, 2001.

Lassaux, S., Renzoni, R., Germanin, A. (2007). Life cycle assessment of water: from the pumping station to the wastewater treatment plant. *International Journal of LCA* **12** (2), 118-126.

Lundie, S., Peter, G.M., Beavis, P.C. (2004). Life Cycle Assessment of wastewater systems: Influence of system boundaries and scale on calculated environmental loads. *Environmental Science and Technology* **34** (1), 180-186.

Hong, J., Hong, J., Otaki, M., Jolliet, O. (2009). Environmental and economic life cycle assessment for sewage sludge treatment processes in Japan. *Waste Management* **29** (2), 696-703.

Nielsen, S. (2003). Sludge treatment in wetland systems. Proceedings of Conference: The Use of Aquatic Macrophytes for Wastewater Treatment in Constructed Wetlands (IWA). 8-10 May. Dias V and Vymazal J. (ed). Lisbon (Portugal).

Nielsen, S. and Willoughby, N. (2005). Sludge treatment and drying reed beds systems in Denmark. Water and Environmental Journal **19** (4), 296-305.

Peruzzi, E., Macci, C., Doni, S., Masciandaro, G., Peruzzi, P., Aiello, M., Caccianti, B. (2007). Wetland cane *(Phragmites australis)* for sewage sludge stabilization. In: *Proceedings of the International Conference on Multi Functions of Wetland Systems,* Padova, Italy. Borin, M. And Bacelle, S. (Eds).

Sarkar, U., Hobbs, S. E. (2003). Landfill odour: assessment of emissions by the flux footprint method. Environmental Modelling & software 18 (2), 155-163.

Uggetti, E., Llorens, E., Pedescoll, A., Ferrer, I., Castellnou, R., García, J. (2009) Sludge dewatering and stabilization in drying reed beds: characterization of three full-scale systems in Catalonia, Spain. Bioresource Technology, 100(17), 3882-3890

Uggetti, E., Ferrer, I., E., Llorens, García, J. (2010). Sludge treated wetlands: A review on the state of the art. *Bioresource Technology* **101** (9), 2905-2912.

Von Sperling, M., Gonçalves, R.F. (2007). Sludge characteristics and production. In: *Sludge Treatment and Disposal*. Cleverson, Von Sperling & Fernandes Eds. IWA Publishing, London, UK, 2007.

Zwara, W. and Obarska-Pempkowiak, H. (2000). Polish experience with sewage sludge utilization in reed beds. Water Science and Technology **41** (1), 65-69